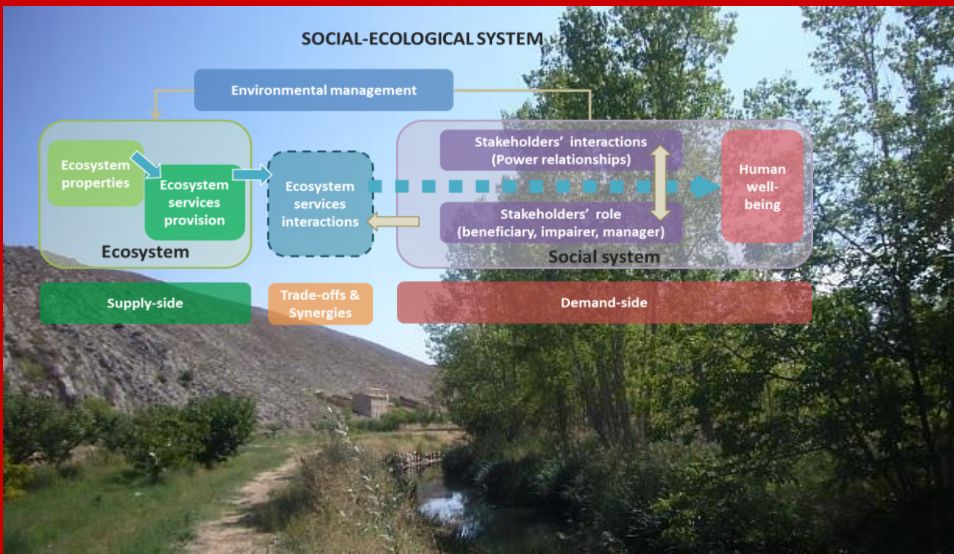
Análisis de las interacciones ecológicas y sociales que intervienen en el flujo de servicios de los ecosistemas. Propuestas para la gestión de la llanura de inundación del río Piedra

Tesis doctoral

Analysis of ecological and social interactions along the flow of ecosystem services. Suggestions for the management of the River Piedra floodplain

PhD. Thesis



María R. Felipe Lucia

Abril 2015

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Memoria presentada por **María R. Felipe Lucia** para optar al grado de Doctora por la Universidad Pablo de Olavide con Mención Doctor Internacional

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Francisco A. Comín Sebastián | F. Javier Escalera Reyes

A mi familia

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RESUMEN

La evaluación de los servicios de los ecosistemas (los beneficios directos e indirectos que los seres humanos obtenemos de los ecosistemas) se ha convertido en una herramienta común y útil para la gestión de los ecosistemas por su conexión directa con las diferentes dimensiones del bienestar humano. Los servicios de los ecosistemas pueden valorarse desde el punto de vista económico (estimando el valor de uso y no-uso de los ecosistemas en términos monetarios), ecológico (midiendo funciones ecológicas o propiedades del ecosistema) o social (basado en los valores que la sociedad atribuye a cada servicio de los ecosistemas). La mayor parte de los estudios que utilizan esta herramienta han utilizado el enfoque económico, mientras que los enfoques ecológico y social han recibido menor atención, por lo que se dispone de métodos menos generalizados para su aplicación. Esta tesis doctoral contribuye a interpretar la relación entre los aspectos ecológicos y sociales que influyen en el flujo de servicios de los ecosistemas y a aplicar el análisis de estas interacciones a la gestión de ecosistemas, tomando como área de estudio la llanura de inundación del río Piedra (cuenca del Ebro, NE España). Las llanuras de inundación de los ríos suelen estar compuestas por un mosaico de usos del suelo que incluye desde bosques de ribera y praderas a cultivos y zonas urbanas o industriales. Encontrar un modelo de gestión sostenible para las llanuras de inundación es especialmente crítico ya que éstas constituyen uno de los hábitats más amenazados y uno de los principales focos de biodiversidad terrestre; mientras que por otra parte, están mayoritariamente dedicadas a la producción agrícola y son el segundo lugar a nivel mundial con mayor interés para el desarrollo urbanístico. Por ello, comprender y evaluar los servicios de las llanuras de inundación, así como incluir estos aspectos en las políticas territoriales y ambientales es fundamental para lograr una provisión equilibrada de servicios a largo plazo.

Respecto a la valoración ecológica de los servicios de las llanuras de inundación, esta tesis doctoral profundiza en el valor que aportan los principales usos del suelo identificados en la llanura de inundación del río Piedra. Para ello, se han estimado los servicios que proporcionan cada uso del suelo y la diversidad vegetal asociada a ellos a partir de datos obtenido mediante muestreos en campo y de datos públicos. Además, se ha estimado la provisión de servicios a tres escalas espaciales: parcela, municipio y paisaje (la llanura de inundación en conjunto). Los datos han sido analizados mediante modelos lineales generalizados y mixtos, tests multi-comparativos y análisis de correlaciones. De esta manera se han identificado, por una parte, los servicios asociados a cada uso del suelo y el efecto del tipo de uso de suelo

en la relación servicios-biodiversidad. Por ejemplo, los hábitats naturales y semi-naturales proporcionaron más servicios y albergaron más diversidad que los usos de suelo cultivados. Además, la mayoría de los índices de diversidad vegetal se correlacionaron positivamente con los servicios de provisión de hábitat y educación ambiental y negativamente con la provisión de alimentos. Por otra parte, se han identificado las sinergias y antagonismos entre servicios de los ecosistemas en los diferentes usos del suelo y escalas espaciales, observando que la escala espacial de análisis determina la estimación de los servicios. Por ejemplo, a escala de parcela, nuestros resultados demuestran que los bosques de ribera proporcionan mayor cantidad de servicios de los ecosistemas, mientras que a escala municipal y de paisaje son los cultivos de cereal los que mayor cantidad de servicios proporcionan debido a que ocupan la mayor extensión de terreno. Además, se propone una clasificación de las interacciones entre servicios de los ecosistemas para identificar las causas de los antagonismos entre servicios de los ecosistemas, según sean éstas de origen biofísico o social (es decir, derivadas de los valores sociales que rigen las decisiones de gestión).

Respecto a la valoración social de los servicios de los ecosistemas, se ha revisado la literatura científica considerando la escala espacio-temporal utilizada, los tipos de participantes involucrados y los métodos empleados. Esta revisión ha constatado que la mayoría de los estudios se llevan a cabo a escala municipal o supra local, los residentes locales sólo están incluidos en un tercio de las evaluaciones, y los métodos más utilizados son la identificación de servicios de los ecosistemas y el orden de preferencias. En base a estos resultados se han desarrollado unas directrices en las que se establecen los puntos fundamentales a incluir en la valoración social de los servicios de los ecosistemas para que los resultados puedan ser comparables y transferibles, y se ha aplicado el modelo propuesto en la valoración social de los servicios de los ecosistemas del valle del río Piedra.

Además, se han explorado las interacciones tanto ecológicas como sociales que intervienen en el flujo de servicios de los ecosistemas al bienestar humano. En la esfera ecológica, se ha identificado mediante un modelo de ecuaciones estructurales que los servicios de soporte y regulación son clave para mantener el flujo de servicios de los ecosistemas a los diferentes agentes sociales de interés. En la esfera social, se han identificado las asimetrías de poder entre agentes de interés que determinan el acceso y la gestión de los servicios de los ecosistemas, mostrando cómo la capacidad de estos agentes para gestionar los servicios de soporte y regulación determina las relaciones de poder entre ellos. La tesis finaliza con un análisis comparativo de la provisión de servicios de los ecosistemas de la llanura de inundación del río Piedra en cinco escenarios alternativos de gestión. La elaboración de escenarios se basó en la combinación de un gradiente de intensidad del uso del suelo y un gradiente de restauración del bosque de ribera. Los resultados señalan que el escenario de conservación del bosque de ribera en combinación con un uso agrario no intensivo proporciona una combinación más equilibrada de servicios.

Las aportaciones de esta tesis doctoral son útiles para integrar el conjunto de servicios que proporcionan los ecosistemas en políticas ambientales y territoriales. De esta manera se pretende fomentar la evaluación de los servicios de los ecosistemas desde múltiples perspectivas, promocionar los paisajes multifuncionales que suministren un conjunto equilibrado de servicios, incluir la participación pública en la toma de decisiones y lograr un acceso más igualitario a los servicios de los ecosistemas.

SUMMARY

The assessment of ecosystem services (the direct and indirect benefits humans receive from ecosystems) has become a common and useful tool in ecosystems management, due to its direct connection to the various dimensions of human well-being. Ecosystem services can be assessed from an ecological, economic, or social approach. The ecological approach focuses on measuring ecological functions or ecosystem properties; the economic approach estimates the use and non-use values of ecosystems in monetary terms; and the social approach is based on the values society attributes to each ecosystem service. Most studies assessing ecosystem services use the economic approach; whereas the ecological and social approaches have received less attention and their methods are still ill-defined. This PhD thesis contributes to understanding the relationships between the ecological and the social aspects that influence the flow of ecosystem services and to applying the analyses of such interactions to ecosystems management through the River Piedra floodplain case study (River Ebro basin, NE Spain). Floodplains are usually a land use mosaic of riparian forests, meadows, agricultural, urban, and industrial areas. Understanding how floodplains can be sustainably managed is especially important given that floodplains are one of the most endangered habitats and biodiversity hotspot, while they are mostly used for agricultural production and are still the second highest worldwide attraction for housing developers. Thus, including the assessment of floodplain ecosystem services in land and environmental policies is key to reaching a balanced supply of ecosystem services in the long term.

Regarding the ecological valuation of ecosystem services in floodplains, this PhD thesis deepens on the value supplied by each land use type identified in the River Piedra floodplain. For this, we assessed ecosystem services supply and estimated plant diversity associated to each land use type. In addition, we estimated ecosystem services supply at three spatial scales: patch, municipality, and landscape (the whole floodplain) using field and public data. Data were analysed using general and mixed lineal models, multi-comparative tests, and correlation analyses. On the one hand, we identified ecosystem services associated to each land use type and the effect of the land use type in ecosystem services-biodiversity interactions. For instance, natural and semi-natural habitats supplied more number of ecosystem services and hosted greater diversity than cultivated land use types. In addition, most plant diversity indexes were positively correlated to habitat provision and environmental education, but negatively correlated to food provision. On the other hand, we identified synergies and trade-offs between ecosystem services across land use types and spatial scales. We found that the spatial

scale at which measurements were taken affected the composition of services. For instance, at patch scale, riparian forest supplied the most services of any land use type, but dry cereal croplands provided the most services across the municipality and landscape because of their large area. Additionally, we propose a classification of ecosystem services interactions that incorporates societal values (as drivers of management decisions) along with biophysical factors as likely causes of ecosystem services trade-offs.

Regarding the social valuation of ecosystem services, we reviewed current trends in literature on spatial-temporal scales, type of participants, and methodology used. We found that most studies are addressed at the municipality or supra-local scale, local residents are included just in a third of the valuations, and the methods most commonly used are both ecosystem services identification and ranking. Based on the agreements which emerged from this review, we proposed a set of guidelines that should be explicit in such assessments to enable comparisons across studies. In addition, we illustrated the proposed framework through the social valuation of ecosystem services in the River Piedra floodplain.

Next, we explored both ecological and social interactions that mediate ecosystem services flow to human well-being. On the ecological side, we identified that regulating and supporting services were key to maintaining the ecosystem services flow to stakeholders using a structural equation model. On the social side, we identified power asymmetries between stakeholders that mediate access and management to ecosystem services. These analyses revealed that the ability of stakeholders to manage supporting and regulating services determine power relationships among them. This PhD thesis concludes with a comparative analysis of ecosystem services supply in the River Piedra floodplain across five alternative management scenarios. Scenarios were based on the combination of a land use intensity gradient and a riparian forest restoration gradient. We found that the scenario fostering riparian forest enhancement and no intensive agricultural use supplied a more balanced set of ecosystem services.

This PhD thesis contributes to provide tools for integrating the assessment of ecosystem services in environmental and land management policies. In doing so, we aim to foster the assessment of ecosystem services from multiple approaches, promote multifunctional landscapes that provide a balanced set of ecosystem services, include public participation in decision-making, and achieve a more equal access to ecosystem services.

CAPÍTULO 1. INTRODUCCIÓN Y MARCO TEÓRICO

Los servicios de los ecosistemas en las llanuras de inundación

Los servicios de los ecosistemas suelen definirse como los beneficios directos o indirectos que los seres humanos obtenemos de los ecosistemas (Millennium Ecosystem Assessment 2005; Evaluación de los Ecosistemas del Milenio 2011). Se clasifican normalmente en cuatro categorías: servicios de abastecimiento, que incluyen los beneficios tangibles o materiales como alimentos, agua y materias primas; servicios culturales, que son beneficios intangibles o inmateriales como el uso recreativo, la relajación, la educación ambiental y el disfrute estético; servicios de regulación, como la regulación del ciclo de nutrientes y del clima y el control de plagas e inundaciones y servicios de soporte, que engloban los mecanismos que sostienen los ecosistemas, como la provisión de hábitat y la formación de suelo (Tabla 1).

El concepto de *servicios de los ecosistemas* constituye una herramienta cada vez más utilizada para la gestión de ecosistemas al incorporar otros aspectos de los ecosistemas que no se valoraban explícitamente hasta ahora, como los servicios de regulación y de soporte y parte de los culturales. Este concepto también contribuye a la sensibilización de la sociedad, dando a conocer al público general nuestra dependencia de la naturaleza y valorando lo que de ella obtenemos gratuitamente. De hecho, uno de los objetivos iniciales y más generales de la valoración de los servicios de los ecosistemas es destacar la contribución de los ecosistemas al bienestar humano (Dasgupta 2001; Millennium Ecosystem Assessment 2005; Carpenter *et al.* 2009).

El enfoque antropocéntrico del concepto de *servicios de los ecosistemas* ha recibido algunas críticas por centrarse en el flujo de servicios dirigidos únicamente al ser humano e ignorar el flujo de servicios que puedan darse entre el resto de componentes de los ecosistemas (Hansson and Wackernagel 1999; Barnaud and Antona 2014a). Sin embargo, los servicios de los ecosistemas permiten conectar directamente con las diferentes dimensiones del bienestar humano (Figura 1), convirtiéndose en una herramienta muy útil para la gestión de ecosistemas (Fisher *et al.* 2009; Bennett *et al.* 2009; de Groot *et al.* 2010; Lamarque *et al.* 2011). En este contexto, surge el concepto de *socio-ecosistema* (Berkes *et al.* 2003; Escalera-Reyes and Ruiz-Ballesteros 2011), que refuerza la idea de la relación interdependiente que la sociedad mantiene con el ecosistema siendo el ser humano el principal beneficiario de servicios.

Tabla 1. Ejemplos de servicios de los ecosistemas agrupados en categorías, indicadores y unidades de medida.

Categoría	Servicio	Indicador	Unidades
Soporte	Estabilidad del suelo	Espesor capa de materia orgánica	cm
	Calidad del hábitat	Riparian Quality Index	-
Regulación	Calidad del agua (depuración del agua)	Nitrito disuelto	ppm
		Nitrato disuelto	ppm
	Formación de suelo	Contenido en materia orgánica	g/100 g
	Regulación de nutrientes	Contenido en carbono	g/100 g
		Contenido en nitrógeno	g/100 g
	Regulación del clima	Variación de la temperatura	°C
	Control biológico de plagas	Estratos de vegetación	nº
	Secuestro de carbono	Secuestro de carbono	CO_2eq
Abastecimiento	Producción de alimentos	Calorías	kcal
		Productividad	kg
	Producción de materias primas	Acumulación de biomasa	T
Cultural	Valor estético	Densidad de fotos	nº/ha
	Recreativo	Densidad de sitios	nº/ha
	Educación ambiental	Densidad de paneles educativos	nº/ha
	Deportes	Densidad de rutas	m/ha
	Disfrute de la naturaleza	Superficie forestal	ha

El estudio de los servicios de los ecosistemas se puede enfocar desde una perspectiva ecológica, económica o social (Figura 2). La aproximación ecológica se centra en medir funciones ecológicas o propiedades de los ecosistemas (de Groot *et al.* 2002), el enfoque económico estima el valor de uso y no-uso de los ecosistemas en términos monetarios (Wilson and Carpenter 1999) y el enfoque social se basa en los valores que la sociedad atribuye a cada servicio (Martín-López *et al.* 2012). La evaluación de las tres aproximaciones proporcionará mayor información de los servicios de los ecosistemas para la gestión del territorio (Oteros-Rozas *et al.* 2012) (Figura 3).

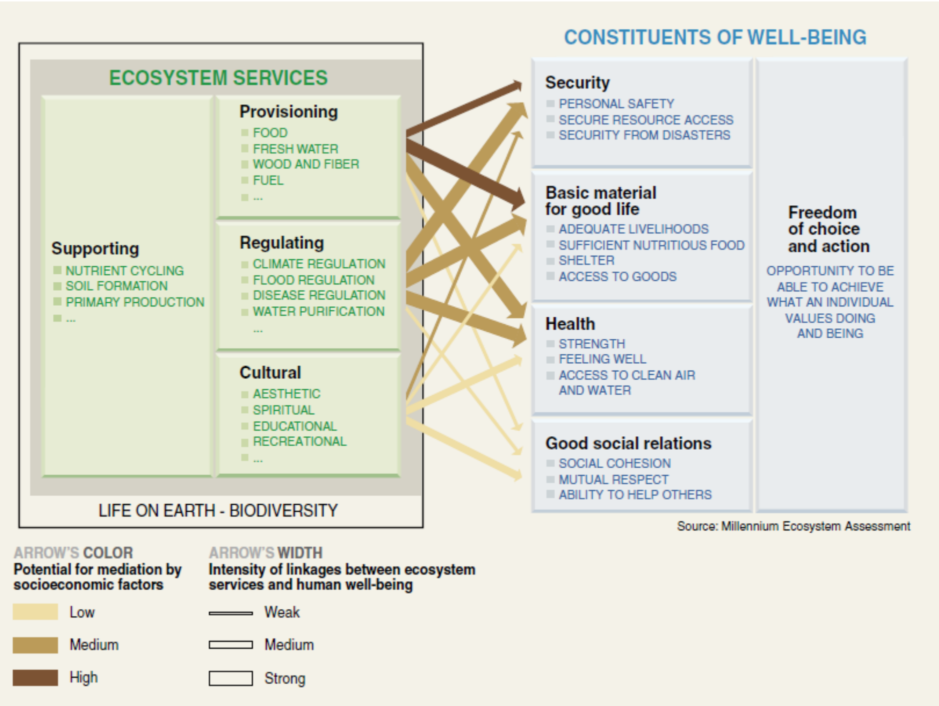

Figura 1. Marco de los servicios de los ecosistemas (panel izquierdo) y su relación con los diferentes constituyentes del bienestar humano (panel derecho). El color de las flechas indica su potencial para ser mediado por factores socioeconómicos y el grosor la intensidad de las conexiones (Millennium Ecosystem Assessment 2005).

Para estimar la provisión de servicios de los ecosistemas, la mayoría de los estudios suelen utilizar Sistemas de Información Geográfica (SIG) e imágenes de satélite (Kreuter *et al.* 2001; Konarska *et al.* 2002; Chen *et al.* 2009), bases de datos (Viglizzo and Frank 2006; Tianhong *et al.* 2010) o programas de modelización (Nelson *et al.* 2009; Goldstein *et al.* 2012). Además, la mayor parte de estos estudios comparan modelos de gestión basados en un único o muy reducido número de usos del suelo (Chan *et al.* 2006; van Oudenhoven *et al.* 2012). Sin embargo, pocos estudios se han basado en la toma de datos locales y en diferentes usos del suelo para estimar los servicios de los ecosistemas (pero véase Raudsepp-Hearne *et al.* 2010), a pesar de que existen evidencias de que la distribución de los usos del suelo afecta a la provisión de servicios (Mitchell *et al.* 2013) y de que estos datos son fundamentales para asegurar una estimación precisa (Nelson *et al.* 2009; Eigenbrod *et al.* 2010). De hecho, el valor total de cada servicio generado en un espacio determinado depende tanto del valor del servicio por unidad de superficie como de la superficie total de cada uso del suelo existente en el área de estudio (Felipe-Lucia *et al.* 2014a).

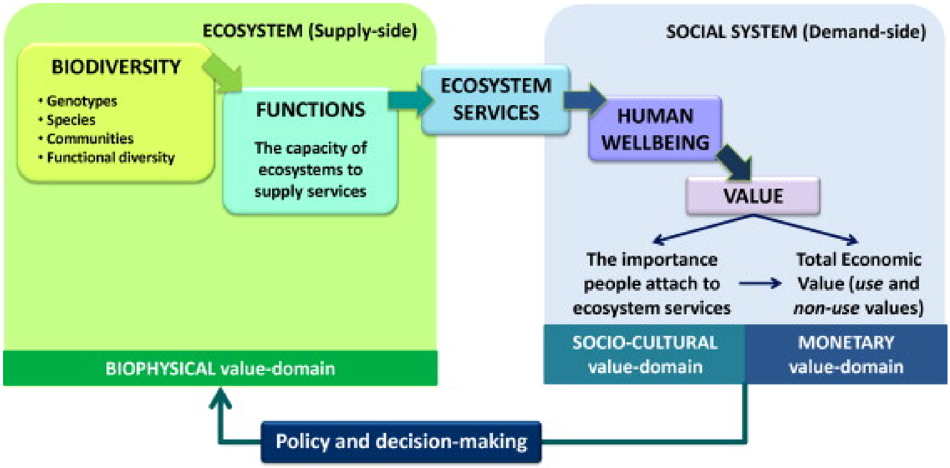

Figura 2. Marco metodológico para evaluar los servicios de los ecosistemas. El marco diferencia la provisión de servicios por parte de los ecosistemas y el uso, disfrute y valor para los usuarios, que pueden valorar los servicios de los ecosistemas desde la perspectiva socio-cultural o monetaria (Martín-López *et al.* 2014).

En esta tesis doctoral, se ha estimado la provisión de servicios de los ecosistemas de una llanura de inundación típica mediterránea diferenciando los principales usos del suelo. La investigación en servicios de los ecosistemas basada en múltiples usos del suelo es escasa (Metzger and Schroeter 2006; Petz and van Oudenhoven 2012) y todavía más rara en las llanuras de inundación (Schindler *et al.* 2013). No obstante, las llanuras de inundación –el espacio contiguo al río inundado durante los períodos de crecida– contribuyen a proveer más del 25% de los servicios de los ecosistemas terrestres (Tockner and Stanford 2002), entre ellos, los asociados con la regulación del agua y del ciclo de nutrientes, la producción de alimentos, los asentamientos humanos y la conservación de la biodiversidad (Posthumus *et al.* 2010; Vidal-Abarca Gutiérrez and Suárez Alonso 2013). Las llanuras de inundación suelen estar compuestas por una matriz de usos del suelo naturales y cultivados. Además, albergan diferentes tipos de ecosistemas (p.ej. bosques de ribera, praderas, matorrales, agroecosistemas, zonas industriales y urbanas) y desempeñan múltiples funciones, por lo que pueden considerarse paisajes multifuncionales (Mander *et al.* 2007). Sin embargo, en la mayor parte de Europa y Norte América, la presión actual sobre las llanuras de inundación para alimentar la creciente población humana está provocando su degradación ambiental a gran escala, causada principalmente por urbanización, deforestación, erosión del suelo, lixiviación de nutrientes, contaminación del agua y captación y desviación de caudales (Simoncini 2008). Por ello, encontrar un modelo adecuado para gestionar las llanuras de inundación es especialmente importante ya que éstas constituyen uno de los hábitats más amenazados y uno de los principales focos de biodiversidad terrestre; mientras que por otra parte, son el segundo lugar con mayor interés para el desarrollo urbanístico a nivel mundial (Moss and Monstadt 2008). Comprender y evaluar los servicios proporcionados por los diferentes usos del suelo de las llanuras de inundación, así como incluir estos aspectos en las políticas territoriales y ambientales es fundamental para lograr una provisión equilibrada de servicios de manera sostenible a largo plazo y la permanencia de la población local.

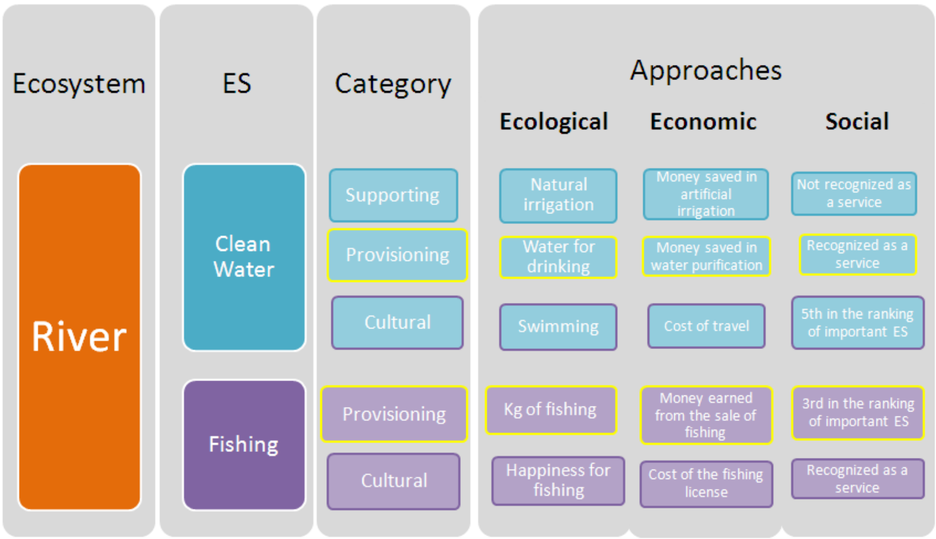

Figura 3. Ejemplo de los enfoques que pueden aplicarse a la evaluación de dos servicios proporcionados por los ecosistemas fluviales. Los servicios de los ecosistemas (ES) se adscriben normalmente a una única categoría (segunda columna, fondo azul para servicios de soporte y fondo lila para servicios culturales). Sin embargo, cada servicio de los ecosistemas puede valorarse por más de una categoría (tercera columna, marco azul para servicios de soporte, amarillo para servicios de abastecimiento y lila para servicios culturales). Dentro de cada categoría, los servicios de los ecosistemas pueden valorarse desde la perspectiva ecológica, económica o social, usando distintos tipos de indicadores (Felipe-Lucia *et al.* 2014b).

Relación entre servicios de los ecosistemas y biodiversidad e importancia de la escala espacial

Los sistemas socio-ecológicos presentan patrones espaciales (p.ej. localización y proporción relativa de los usos del suelo, biodiversidad, intereses de los agentes sociales, responsabilidades institucionales; Hein *et al.* 2006; Concepción *et al.* 2012). Estos patrones pueden variar a diferentes escalas espaciales (p.ej. a nivel de parcela, municipio o paisaje en general) influyendo en las funciones ecológicas (Pringle *et al.* 2010), la conservación de los ecosistemas y el bienestar humano (DeFries *et al.* 2004). Utilizar datos locales de calidad y análisis multi-escalares son fundamentales para diseñar planes de gestión adecuados, comprender las contrapartidas que conllevan y facilitar la toma de decisiones (Carpenter *et al.* 2009). Además, los servicios de los ecosistemas no actúan de manera independiente (Bennett *et al.* 2009), sino que interaccionan entre ellos creando antagonismos y sinergias diferentes según la escala espacial (Felipe-Lucia *et al.* 2014a). A pesar de esta evidencia, la escala más apropiada para gestionar cada servicio todavía se desconoce (Hein *et al.* 2006). Por ello, aprender a gestionar las llanuras de inundación a diferentes escalas espaciales para proporcionar múltiples servicios permitirá mejorar la provisión de servicios de los ecosistemas al conjunto de la sociedad.

Estudios recientes han revelado múltiples sinergias entre las medidas de gestión destinadas a incrementar los servicios de los ecosistemas y la probabilidad de mejorar la conservación de la biodiversidad (véase la revisión de Cimon-Morin et al. 2013). Por ejemplo, determinadas prácticas de gestión de agroecosistemas han permitido conservar la biodiversidad sin perder significativamente producción de alimentos (Polasky *et al.* 2005; Scherr and McNeely 2008; Sayer *et al.* 2013). Aunque la gestión de los agroecosistemas para favorecer múltiples servicios no suele lograr el 100% de conservación de la biodiversidad (Macfadyen *et al.* 2012), la pérdida de biodiversidad se considera pequeña (Chan *et al.* 2006; Nelson *et al.* 2009). A escala europea, las políticas agroambientales para mejorar la biodiversidad de zonas agrícolas han mejorado considerablemente tanto la biodiversidad como la provisión de servicios de los ecosistemas (Whittingham 2011; Ekroos *et al.* 2014). Estas políticas de gestión son fundamentales, ya que la pérdida de biodiversidad amenaza la provisión de servicios de los ecosistemas (Balvanera *et al.* 2006; Meli *et al.* 2014) y el bienestar humano (Díaz *et al.* 2006).

Por ello, investigadores, gestores de ecosistemas y la sociedad en general demandan cada vez más políticas que promuevan la provisión de servicios de los ecosistemas conservando la biodiversidad asociada a ellos (Turner *et al.* 2007; Haines-Young and Potschin 2010). Gestionar los ecosistemas para alcanzar ambos objetivos requiere analizar tanto las sinergias como los compromisos entre conservación y esfuerzo económico (Naidoo and Ricketts 2006; Tallis *et al.* 2008). Recientemente, la escasez de financiación está llevando a proteger más lugares importantes para la provisión de servicios de los ecosistemas que espacios para la conservación de la biodiversidad (Goldman *et al.* 2008; Cimon-Morin *et al.* 2013). Sin embargo, existe la posibilidad de lograr ambos objetivos protegiendo la biodiversidad no solamente en áreas naturales sino también en zonas intensamente ocupadas por usos humanos que proporcionen múltiples servicios (Goldman *et al.* 2008; Kareiva and Marvier 2012), como las llanuras de inundación de los ríos. Establecer políticas para gestionar este tipo de ecosistemas de manera que satisfagan las necesidades del ser humano a largo plazo sin poner en peligro la conservación de la biodiversidad es actualmente uno de los mayores retos tanto para la ecología como para la gestión del territorio y el desarrollo de la sociedad humana.

Valoración social de los servicios de los ecosistemas

La gestión del territorio suele basarse exclusivamente en datos económicos o ecológicos ignorando la opinión y valores de la sociedad al respecto (Bryan *et al.* 2010; Satterfield *et al.* 2013). En políticas relacionadas con la gestión del agua se requiere consultar a la sociedad implicada (Directiva Marco del Agua, art. 48); sin embargo, en políticas relacionadas con la gestión de los ecosistemas todavía no es así. Afortunadamente, la evaluación de los servicios de los ecosistemas está empezando a incorporar la valoración social, junto con la ecológica y la económica, en la valoración conjunta de los servicios de los ecosistemas (Ronnback *et al.* 2007; Cowling *et al.* 2008; Paetzold *et al.* 2010; Oteros-Rozas *et al.* 2012; Martín-López *et al.* 2014; Zorrilla-Miras *et al.* 2014). Este esfuerzo es especialmente valioso cuando el objetivo del estudio es aportar información para mejorar la calidad de vida y el bienestar humano mediante la gestión de los ecosistemas.

Sin embargo, la metodología existente para evaluar los servicios de los ecosistemas desde la perspectiva social es todavía confusa (Menzel and Teng 2010), lo que reduce su potencial para asesorar políticas ambientales y territoriales (Chan *et al.* 2012). Por ejemplo, la diferencia entre el enfoque económico y el social no está claro, lo que conlleva al frecuente uso de métodos econométricos para evaluar las preferencias sociales de los servicios de los ecosistemas. En otros casos, el enfoque social sólo se implementa para valorar los servicios de tipo cultural, obviando el resto de servicios de los ecosistemas (regulación, soporte y abastecimiento) (Newton *et al.* 2012). La omisión de los otros tipos de servicios en la valoración social de los servicios de los ecosistemas puede ser debido, entre otras causas, a la cantidad de tiempo y experiencia que dichos métodos requieren, y a la frecuente confusión entre la evaluación de la categoría de servicios "*culturales*" (es decir, los beneficios inmateriales que obtenemos mediante crecimiento espiritual, desarrollo cognitivo, reflexión, recreación y experiencias estéticas; Millennium Ecosystem Assessment 2005), y la valoración del conjunto de los servicios de los ecosistemas (incluyendo todas las categorías) por parte de la sociedad.

En la gestión de ecosistemas, la valoración social se ha implementado fundamentalmente con el fin de lograr objetivos políticos (p.ej. en la gestión de recursos hídricos y naturales; Menzel and Teng 2010). Sin embargo, incrementando la participación pública en la valoración social de los servicios de los ecosistemas (Chan *et al.* 2012) se podrían legitimar las políticas de gestión y obtener decisiones satisfactorias para un mayor número de agentes sociales de interés, como habitantes locales, administraciones y conservacionistas (Menzel and Teng 2010). Desarrollar un marco que guíe la valoración social de los servicios de los ecosistemas es un reto que requiere la colaboración de las ciencias naturales y sociales (Maass *et al.* 2005; Raymond *et al.* 2013).

Las relaciones de poder en el flujo de servicios de los ecosistemas

El marco de los servicios de los ecosistemas ha permitido al público en general reconocer los beneficios que la naturaleza proporciona a las personas (Costanza *et al.* 2014). Sin embargo, no todas las personas disfrutan de estos servicios por igual. Investigaciones recientes señalan que las características geográficas y espaciales propician desigualdades en la provisión de servicios de los ecosistemas (Hein *et al.* 2006; Fisher *et al.* 2009). Por ejemplo, mientras que las poblaciones en la cabecera de los ríos suelen disfrutar de la calidad del agua de un río, probablemente las poblaciones situadas más abajo no la reciban. No obstante, el potencial que tienen los ecosistemas para beneficiar a las personas no depende únicamente de las características espaciales del flujo de servicios (Chan *et al.* 2006; Naidoo and Ricketts 2006; Martín-López *et al.* 2009; Bagstad *et al.* 2014), sino de las múltiples interacciones entre servicios de los ecosistemas (Rodriguez *et al.* 2006). Por una parte, éstas dependen de las propias interacciones entre los componentes biofísicos de los ecosistemas (Villa *et al.* 2014b) y de las interacciones entre servicios de los ecosistemas que crean sinergias y antagonismos (Bennett *et al.* 2009). Por otra parte, las interacciones entre los agentes de interés, causadas en parte por relaciones de poder, pueden determinar el acceso a los servicios de los ecosistemas y su gestión.

El concepto *relaciones de poder* es muy utilizado en la gestión de recursos naturales para determinar asimetrías en el acceso a los recursos (Ribot and Peluso 2003; Raik *et al.* 2008; Reed *et al.* 2009; Crona and Bodin 2010; Akbulut and Soylu 2012; Barnaud and Van Paassen 2013). En las ciencias sociales, este concepto se utiliza para destacar las asimetrías consustanciales a las relaciones sociales (Emerson 1962; Stone 1988; Foucault 1988; Gliscynski 1989; Escalera-Reyes and Ruiz-Ballesteros 2011). Por ejemplo, la antropología ecológica y la ecología política ya incorporan el concepto de poder en las interacciones entre el ser humano y el medio ambiente (Fabinyi *et al.*

2014). En la literatura sobre servicios de los ecosistemas, sólo se han desarrollado estudios que analizan las relaciones de poder en el contexto de pagos por servicios de los ecosistemas (Corbera *et al.* 2007; Vatn 2010; Muradian *et al.* 2013; Pascual *et al.* 2014), pero todavía no se ha estudiado el efecto de las relaciones de poder en el acceso y uso de los servicios de los ecosistemas ni en las interacciones entre los propios servicios. Las asimetrías de poder entre agentes o grupos sociales de interés conllevan que unos puedan usar un servicio o un conjunto de servicios de los ecosistemas determinado mientras otros quedan excluidos. Por tanto, las asimetrías de poder pueden crear conflictos sociales (Turner *et al.* 2003; Hein *et al.* 2006) y afectar al bienestar humano (Daw *et al.* 2011). Por ejemplo, los agentes de interés con poder pueden decidir cuáles son los servicios disponibles y regular el acceso a ellos, afectando negativamente a agentes no empoderados al reducir su capacidad para acceder a los servicios de los ecosistemas. Además, en las decisiones de gestión subyacen relaciones de poder que modifican las interacciones entre servicios de los ecosistemas, resultando en antagonismos y pérdida de servicios (Rodriguez *et al.* 2006; Felipe-Lucia *et al.* 2014a).

El estudio de las relaciones de poder contribuye a mostrar lagunas existentes entre la producción de servicios por un ecosistema y los beneficios reales que las personas reciben. Dichas lagunas permiten identificar a aquellas personas dependientes de ciertos servicios de los ecosistemas que están en riesgo de quedar excluidas del acceso a éstos (Daw *et al.* 2011). Además, en la gestión de los servicios de los ecosistemas todavía no se consideran las relaciones de poder, los contribuyentes a la producción de servicios, los beneficiarios, los que deterioran los servicios y aquellos que quedan excluidos (es decir, los "*perdedores*"; Harrington *et al.* 2010) (Barnaud and Antona 2014a). Las relaciones de poder emergen como un factor clave que influye en el acceso de las personas a los servicios de los ecosistemas, en las interacciones entre agentes sociales de interés, y en la gestión del medioambiente que determina la provisión de servicios de los ecosistemas. Por tanto, integrar las relaciones de poder en la investigación sobre servicios de los ecosistemas constituye un punto fundamental para la gestión de los socio-ecosistemas (Figura 4).

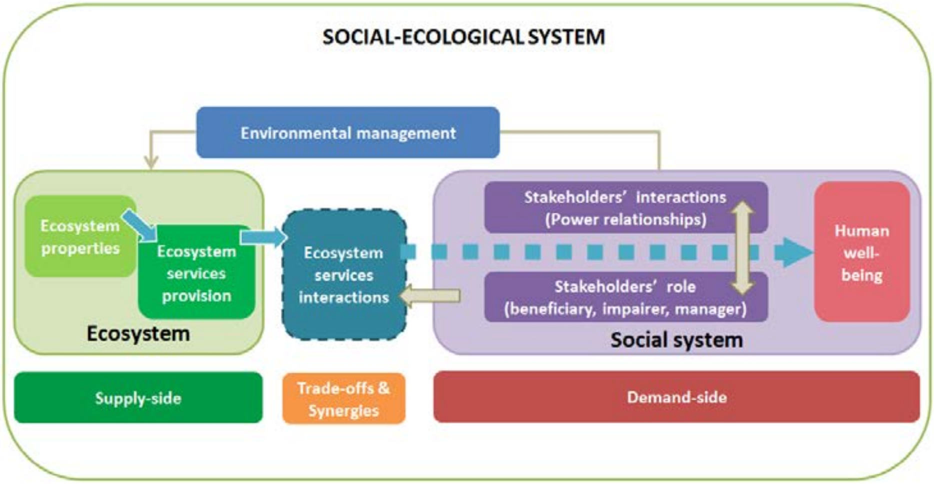

Figura 4. Marco conceptual de las interacciones a lo largo del flujo de servicios de los ecosistemas desde su provisión a su uso y al bienestar humano. El marco destaca las interacciones entre servicios de los ecosistemas y entre agentes sociales de interés que intervienen y pueden modificar el acceso de las personas a los servicios de los ecosistemas. Las flechas azules representan el flujo de servicios de los ecosistemas y las flechas beige denotan interacciones que ocurren entre los componentes sociales del socio-ecosistema o que surgen de éstos (a partir de Haines-Young and Potschin 2010; Martín-López et al. 2014; Spangenberg et al. 2014).

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CAPÍTULO 2. OBJETIVOS

El objetivo general de esta tesis doctoral es identificar y evaluar las interacciones ecológicas y sociales que intervienen en el flujo de servicios de los ecosistemas al bienestar humano a partir de la evaluación de los servicios de los ecosistemas desde las perspectivas ecológica y social, considerando los diferentes usos del suelo de una llanura de inundación a través de diversas escalas espaciales, y aplicar estas perspectivas a la gestión del socio-ecosistema.

Los objetivos específicos de esta tesis doctoral son:

1. Identificar la relación entre servicios de los ecosistemas y biodiversidad en los diferentes usos del suelo de una llanura de inundación típica mediterránea (Felipe-Lucia and Comín 2015; Capítulo 3).

2. Evaluar la provisión de servicios de los ecosistemas de una llanura de inundación a diferentes escalas espaciales, las sinergias y compromisos existentes, y el efecto del cambio del uso del suelo en la provisión de servicios de los ecosistemas (Felipe-Lucia *et al.* 2014a; Capítulo 4).

3. Revisar cómo la valoración social de los servicios de los ecosistemas ha sido aplicada en la literatura científica para proponer unas directrices que guíen este tipo de evaluaciones (Felipe-Lucia *et al.* 2014b; Capítulo 5).

4. Exponer el papel de las interacciones tanto ecológicas como sociales (relaciones de poder) que intervienen en el flujo de servicios de los ecosistemas desde la provisión a los usuarios y el bienestar humano y plantear un marco para la integración de ambas (Felipe-Lucia *et al.* [*en revisión*]; Capítulo 6).

5. Aplicar la valoración de los servicios de los ecosistemas a la elaboración de escenarios de futuro para la gestión de la llanura de inundación del río Piedra (Felipe-Lucia and Comín [*en preparación*]; Capítulo 7).

Nota: Esta tesis doctoral se presenta por compendio de artículos, por lo tanto, el estilo y formato de referencias de cada artículo se ha mantenido lo más fiel posible al de la revista correspondiente.

OBJECTIVES

The general aim of this PhD thesis is to identify and to assess the ecological and social interactions that mediate ecosystem services flows, based on the assessment of ecosystem services from both the ecological and the social approach in a floodplain agroecosystem across multiple land use types and spatial scales, and to apply such insights to the management of the socio-ecosystem.

Specifically, this PhD thesis aimed to:

1. *Disentangle the relationships between ecosystem services and biodiversity in a typical Mediterranean floodplain agroecosystem by disaggregating its land use types (Felipe-Lucia and Comín 2015; Chapter 3).*

2. *Assess: i) the ecosystem services supply in a floodplain across spatial scales, ii) the trade-offs and synergies arisen, iii) how land use change might affect the provision of ecosystem services (Felipe-Lucia et al. 2014a; Chapter 4).*

3. *Explore how the social valuation of ecosystem services has been addressed to date in the scientific literature and to propose a set of guidelines to undertake such assessments (Felipe-Lucia et al, 2014b; Chapter 5).*

4. *Expose the role of both ecological and social interactions (power relationships) that mediate ecosystem services flows from supply to users, and to propose a conceptual framework to integrate such information (Felipe-Lucia et al. [under review]; Chapter 6).*

5. *Apply the assessment of ecosystem services to scenario planning for the management of the River Piedra floodplain (Felipe-Lucia and Comín [in prep.]; Chapter 7).*

Note: This PhD thesis is presented as a compendium of articles. Therefore, the style and citations formats have been kept as accurate as possible to the respective journal.

CAPÍTULO 3. ECOSYSTEM SERVICES-BIODIVERSITY RELATIONSHIPS DEPEND ON LAND USE TYPE IN FLOODPLAIN AGROECOSYSTEMS*

ABSTRACT. Managing agricultural floodplains to meet present and future human requirements without jeopardizing biodiversity conservation is a challenge for land use planners and ecologists. This paper aims to disentangle the relationships between ecosystem services and biodiversity in multifunctional landscapes, such as floodplain agroecosystems, by disaggregating their values across land use types. We measured eight ecosystem services (gas regulation, soil formation, nutrient regulation, habitat provision, food provision, raw materials production, education, and recreation) and six plant diversity indexes (richness, abundance, and true diversity for both plant species and growth forms) in seven land use types identified in the floodplain of the River Piedra (Spain). We observed that all land use types provided services to some extent, but each one was better at providing certain services. Natural or semi-natural habitats provided more services and hosted greater diversity than cultivated land use types. In addition, five diversity indexes were strongly correlated to at least three ecosystem services each one. Habitat provision and education were the ecosystem services positively correlating to most diversity indexes, whereas food provision was negatively correlated to all diversity indexes. Moreover, analyzing the interactions between ecosystem services and biodiversity across land use types, we observed that land use type was the controlling factor regarding the sign and significance of the interaction. The results of this study suggest that, in floodplains agroecosystems, a mosaic landscape of different land use types helps support ecosystem services and contributes to maintaining biodiversity while using local resources. Such land use policies might manage agricultural floodplains at the landscape scale while still being able to accommodate specific measures for each land use type. Moreover, riparian forests should be preserved and restored across the floodplain as they are hot spots for biodiversity and ecosystem services provision.

Key words: Ecosystem services; Plant diversity; Land management policies; Riparian habitat; Floodplain; Multifunctional.

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Introduction

Policies to promote ecosystem services supply while safeguarding biodiversity conservation are increasingly demanded by researchers, land managers, and society (Turner et al., 2007; Haines-Young and Potschin, 2010). Managing to reach both goals requires analyzing trade-offs and synergies between conservation and economic efforts (Naidoo and Ricketts, 2006; Tallis et al., 2008). Recent studies revealed cost-effective management schemes to enhance ecosystem services provision while protecting biodiversity. For instance, specific management practices in agricultural landscapes enabled biodiversity conservation without significant loss in provisioning services such as food production (Polasky et al., 2005; Scherr and McNeely, 2008; Sayer et al., 2013). In fact, agri-environmental schemes enhancing farmland biodiversity have been argued to deliver substantial benefits to both biodiversity and ecosystem services provision (Ekroos et al., 2014) and have been evidenced in Europe (Whittingham, 2011). Although some management plans to enhance ecosystem services failed to achieve biodiversity conservation goals (Chan et al., 2006, 2011; Macfadyen et al., 2012), there are many synergies between management policies enhancing ecosystem services and the likely to increase overall biodiversity conservation (see Cimon-Morin et al., 2013 for a review) with little trade-offs (Nelson et al., 2009). Despite the controversy, the scientific community agrees that the loss of biodiversity endangers ecosystem service provision (Balvanera et al., 2006; Meli et al., 2014) and human well-being (Díaz et al., 2006).

Funding scarcity is increasingly leading to protect important sites for ecosystem services conservation over highly biodiverse sites (Goldman et al., 2008; Cimon-Morin et al., 2013). However, conservationists are stressing the possibility of supporting both targets by protecting biodiversity not only in natural areas but also in human-dominated areas providing many ecosystem services (Goldman et al., 2008; Kareiva and Marvier, 2012), such as agroecosystems. Many agricultural landscapes are successfully managed to enhance multiple ecosystem services while protecting biodiversity (De Groot, 2006; Fischer et al., 2006; Scherr and McNeely, 2008; Anton et al., 2010). Such studies converged in promoting policies encouraging multifunctional landscapes (Mander et al., 2007), which are composed of a matrix of natural and managed land uses providing bundles of ecosystem services (Bennett and Balvanera, 2007; Lovell and Johnston, 2008).

Most floodplains are usually composed of a mosaic of land uses including natural habitats and cultivated lands and can be considered multifunctional landscapes since they provide many services associated to water regulation, nutrient cycling, food provision, human settlement, and biodiversity conservation, among others (Tockner and Stanford, 2002; Posthumus et al., 2010). Hence, establishing policies to manage floodplains to meet long-term human requirements without jeopardizing biodiversity conservation is challenging. Previous studies focused on comparing management schemes of single land uses, but research regarding ecosystem services and biodiversity across multiple land use types is scarce (e.g. Metzger et al., 2006; Petz and van Oudenhoven, 2012) and still rare in floodplains (Schindler et al., 2013). Understanding and evaluating the ecosystem services supplied by the different land use types of floodplains and their linkages with associated biodiversity, will help policy makers to incorporate this issue into sustainable land management policies.

This paper aims to disentangle the relationships between ecosystem services and biodiversity in a floodplain agroecosystem by disaggregating its land use types. We hypothesized that natural land use types deliver a greater number of ecosystem services and host greater biodiversity than cultivated ones, but that many synergies are feasible. To test these hypotheses, we firstly assessed a set of ecosystem services provided by the most common land uses in a floodplain. Secondly, we estimated plant diversity in each land use type using different indexes. Thirdly, we analyzed the relationships between ecosystem services and plant diversity across land use types. Finally, we suggest practical recommendations for policies aiming to enhance both biodiversity and ecosystem services in floodplain agroecosystems.

Methods

Study area

The study area is the floodplain of the River Piedra, which is located in north-east Spain (Fig. 1a). The annual average temperature is 12.7 ºC and the annual average rainfall is 450 mm. River Piedra is 76 km long and its floodplain occupies 19.3 km^2, ranging in width from 50 to 300 m. The area of the River Piedra watershed is 923 km^2, ranging from 1100 m.a.s.l. down to 600 m.a.s.l. in the river mouth at the River Jalon, which is a major affluent of the River Ebro in its right south margin. The dominant land use of River Piedra floodplain is agriculture (46.6%; Table 1), including 9.3% irrigated cereal crops (IC), 27.8% dry cereal crops (DC), 5.8% fruit groves (FG), 3.7% poplar groves (PG), and 1.5% crops in different abandonment stages (AC). In this study abandoned crops were semi-natural areas, as they remain unmanaged for over five years. Natural areas (i.e. riparian forests, RF) occupy 2.6% and urban areas (UA) 6.3%. Minor land uses, including vineyards and almond trees among others, covered less than 1% of the total floodplain area each of them. Additionally, the watershed comprises a reservoir of 5.60 km^2 surface.

Table 1. Main land use types in the floodplain of the River Piedra, extension, and cover percentage. Others refer to minor land uses covering less than 1% each of these (e.g. vineyards, almond trees).

Main land use types	Abbrev.	Extension (km²)	Percentage
Abandoned crops	AC	0.28	1.45
Dry cereal crops	DC	5.38	27.82
Fruit groves	FG	1.12	5.81
Irrigated crops	IC	1.80	9.30
Poplarg	PG	0.71	3.66
Riparian forests	RF	0.51	2.64
Urban areas	UA	1.23	6.34
Others	O	1.20	6.23
Total study area		**12.23**	**100**

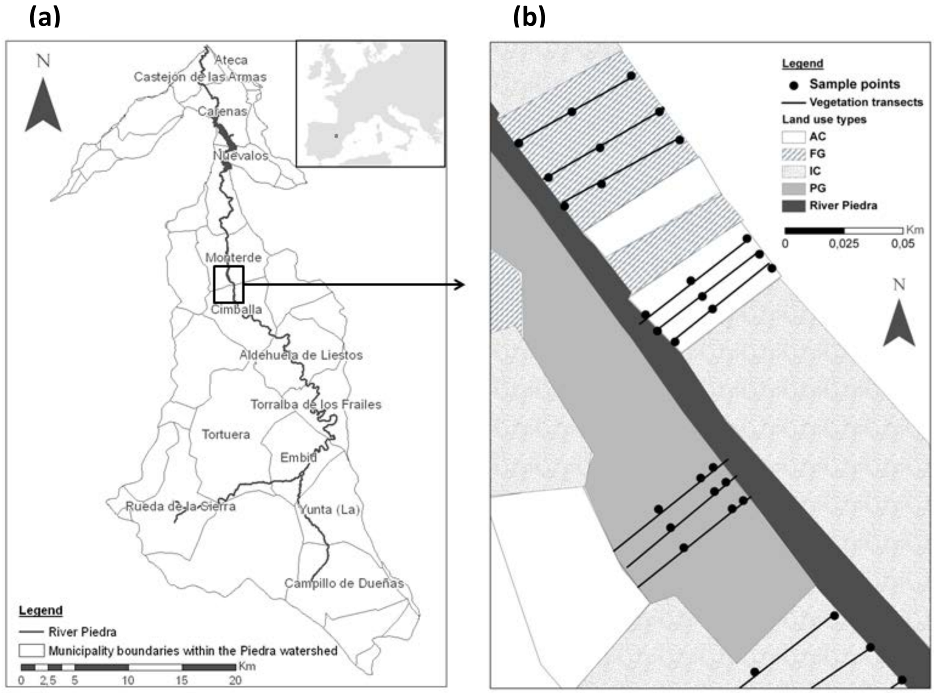

Figure 1. The watershed of River Piedra in NE Spain divided in municipality boundaries (a); and example of sampling design in a stretch of the river (b).

Data collection

The identification of the main land uses of the River Piedra floodplain was based on the Spanish crop and land use digital map at scale 1:50 000 (MARM 2009) and validated through field observations. The coverage of each land use type was calculated using ArcGIS 10 (ESRI 2012).

<u>Ecosystem services assessment</u>

Among the main services delivered by floodplains identified at European scale (Harrison et al., 2010) and at national scale (Vidal-Abarca and Suárez, 2013), we selected a set of eight expected to show different values among the land use types of a typical floodplain agroecosystem. The selection was based on our ability to collect data and included regulating (3), supporting (1), provisioning (2), and cultural (2) services (MA 2005) (Table 2). Data sources and methods used to evaluate ecosystem service provision across land use types are detailed below.

Gas regulation. We used CO2 sequestration as a surrogate of gas regulation (Trabucchi et al., 2014). Annual CO2 sequestration rates by land use type were obtained from a national database (Montero et al., 2005; CITA, 2008) which estimated the amounts of carbon stored by above- and below-ground biomass of the main Spanish plant species and woody formations. Calculations are based on the species annual growth and transformed into CO2 equivalent tons per hectare using stoichiometric equations (Montero et al., 2005). We used data from the closest plant species or woody formations to the land cover composition of our study area (i.e. poplar groves for PG, riparian forests for RF; and average data of apple, pear, peach and plum groves for FG). Herbaceous species – and therefore, IC and DC – are not included because their annual CO2 storage balance is null (CITA, 2008); for AC, only its woody formations (e.g. hawthorn) have been counted. Urban areas have not been included either, since they act usually as a source of carbon rather than as a sink (but see Davies et al., 2011).

Soil formation. We recorded the organic matter layer thickness in topsoil (0-10 cm) as an indicator of this service (Daily et al., 1997). We sampled three patch replicates by land use type except in urban areas, where most of the soils are sealed. Three transects perpendicular to the river channel were established in each patch and three measurements along each transect were taken at 1 m, 5 m, and 15 m away from the river (see Fig. 1b). The organic matter layer depth (cm), excluding leaf litter, was recorded in the field with a measuring tape in September 2010, July 2011, and July 2012, and average data of each point across the three years were used as an indicator.

Nutrient regulation. We measured the organic matter content in topsoil (0- 10 cm) as an indicator of this service (Daily et al., 1997) following the same field sample design described above for soil formation. Half a kilogram of topsoil was collected at each point, dried (48 h at 60ºC), sieved and milled. Total organic matter was analyzed using the LOI protocol (Lost On Ignition, Nelson and Sommers, 1996) and the average value (as soil weight percentage) of the three years was used for each sampling point.

Habitat provision. To assess the provision of habitat by land use type we used the Riparian Quality Index (RQI) (González del Tánago and García de Jalón, 2011). A good riparian condition is related to terrestrial diversity (Grant and Bennett, 2006). Additionally, habitat provision by riparian areas is especially important where adjacent land has been cleared or modified (Martin et al., 2006), such as in agricultural landscapes. The RQI evaluates seven riverbank attributes: (i) dimensions of land with riparian vegetation (average width of riparian corridor); (ii) longitudinal continuity, coverage and distribution pattern of riparian corridor (woody vegetation); (iii) composition and structure of riparian vegetation; (iv) age diversity and natural regeneration of woody species; (v) bank conditions; (vi) floods and lateral connectivity; and (vii) substratum and vertical connectivity). The RQI provides a relative score between 10 and 120 that was reclassified from 0 (habitat provision extremely low) to 100 (habitat provision extremely high). The RQI was estimated in three plot replicates by land use type between July 2010, July 2011, and July 2012.

Food provision. We estimated the average yield (kilograms per hectare) of each land use type from the latest update of a national public database (INE, 2008). We averaged irrigated wheat, barley and corn yields to estimate food provision by IC; dry wheat, barley and corn yields for DC; and apple, pear, peach and plum grove yields for FG. The other land uses were assigned a yield value of 0.

Raw materials production. We used the yearly aboveground dry biomass accumulation by land use type as a measure of the raw materials production. Values were obtained from a national database (Montero et al., 2005; CITA, 2008) that estimated the annual growth rates of woody species as tons of dry biomass per hectare, according to the average timber diameter. We adapted data from the closest woody species to the land cover composition of our study area (i.e. poplar groves for PG, riparian forests for RF; and average data of apple, pear, peach and plum groves for FG). Herbaceous species – and therefore, IC and DC – are not included because their annual accumulated biomass balance is null (CITA, 2008), whereas for AC, only its woody formations (e.g. hawthorn) have been counted. Note that biomass production is an indicator of the potential biomass provision by each land use type, thus referring to the potential use of the biomass as a raw material (i.e. making this use of the land incompatible with the provision of other services such as fruit production or gas regulation).

Education. We used the number of educative panels with information highlighting the importance of the ecosystem as an indicator for this service. This was the only available indicator distinguishing among land use types. Educative panels were counted in each municipality by land use type in August 2012. In order to perform the correlation analysis at a consistent spatial scale (namely, patch), these data were transformed into a density measure (i.e. total number of items by land use type and municipality/land use type cover extent at each municipality).

Recreation. We used the number of areas used for social amenity (e.g. picnic areas) within the study area as an indicator for this service (Posthumus et al., 2010) as this was the only available recreational attribute distinguishing among land use types. The number of areas in each municipality was counted by land use type in August 2012. In order to perform the correlation analysis at a consistent spatial scale (namely, patch), these data were transformed into a density measure (i.e. total number of items by land use type and municipality/land use type cover extent at each municipality).

Plant diversity surveys

We used plant diversity to quantify floodplain biodiversity (Isbell et al., 2011; Maestre et al., 2012) as high diversity has been related to high ecosystem function (Tilman et al., 2001; Balvanera et al., 2006), and increasing riparian plant diversity has been related to greater riparian wildlife and biodiversity (Meli et al., 2014). We measured plant diversity by surveying three plot replicates per land use type in July 2012. Urban areas were excluded as soils were sealed. Within each plot, three floodplain-wide transects (average transect length 57 m) perpendicular to the river channel were established 25 m apart. In each transect, we used the point-intercept method (Goodall, 1952) every 10 cm to estimate species occurrence and percent covers of each plant species (i.e. number of contacts relative to the total number of points sampled). Identification of plants at the genus or species level was corroborated using a regional herbarium (namely, herbarium of Jaca: http://proyectos.ipe.csic.es/herbario) and a botanist expert.

We calculated three commonly used diversity indexes that we called the taxonomic approach: species richness (i.e. number of species; SR); total abundance (i.e. species cover; SA), and true species diversity (i.e. exponential of Shannon entropy; TSD) (Shannon and Weaver, 1949). TSD is considered a “true diversity index” because it provides consistent conservation conclusions (Jost et al., 2010), i.e. it avoids mathematical bias caused by very common or uncommon species, and remains representative of the number of species and the distribution of species abundance in the community. Additionally, we classified vegetation records into four growth-forms (namely, herb, creeper, shrub, and tree) and estimated the same three diversity indexes: growth-forms richness (GR), growth-forms total abundance (GA), and true growth-forms diversity (TGD) that were so-called the functional approach. All indexes were calculated using the vegan package (Oksanen et al., 2013) of the R software (R Development Core Team, 2013).

Table 2. Ecosystem services studied in the floodplain of the River Piedra. (Abbreviations: OM, Organic Matter; RQI, Riparian Quality Index).

ES Group	ES name	ES indicator	Units
Regulating	Gas regulation	CO_2 sequestration	CO_2 eq. t/ha · year
Regulating	Soil formation	OM layer	cm
Regulating	Nutrient regulation	OM content	Weight percentage
Supporting	Habitat provision	RQI	Scoring
Provisioning	Food provision	Yield	kg/ha · year
Provisioning	Raw materials production	Biomass production	t/ha · year
Cultural	Education	Educative panels	Number of items
Cultural	Recreation	Recreational sites	Number of items

Statistical analyses

To determine significant differences in the provision of ecosystem services and plant diversity among land use types we used linear models (LM) and multiple comparison tests (multcomp R package, Hothorn et al., 2008). Models were fitted according to the data structure, so to fit data to a Poisson distribution we applied generalized linear models (GLM); to allow nested data structures (Zuur et al., 2009) we used generalized linear mixed models (GLMM) controlling the residual dispersion of the distance to the river through their random effects. GLMM were fitted to a Gaussian or Poisson distribution as necessary, using respectively the nlme (Pinheiro et al., 2013) and the lme4 (Bates et al., 2012) R packages. To explore the linkages between ecosystem services and plant diversity values we followed the approach of Raudsepp-Hearne et al. (2010) using the Pearson correlation coefficient (r). P values and Pearson coefficients were obtained from the Hmisc R package (Harrell, 2014).

Results

Ecosystem services provided by each land use

Riparian forests and fruit groves were the land use types providing the greatest number of ecosystem services (7 of the 8 services studied) in our study area. Poplar groves provided six services, abandoned crops five services, dry and irrigated cereal crops four services, and urban areas supplied the least (3 services). In addition, each land use type was better at providing certain services. For instance, the services mainly provided by riparian forests were soil formation, nutrient regulation, habitat provision, education and recreation; fruit groves were the best at supplying gas regulation, food provision and raw materials; dry cereals crops did better in nutrient regulation; irrigated cereal crops in food provision; abandoned crops and poplar groves in soil formation; and urban areas provided also the most recreation (Fig. 2a, see averaged values in Table A.1). Below we detail the provision of ecosystem services by land use type. P-values and significant differences between land uses are detailed in Table A.2.

Gas regulation: fruit groves, riparian forests and poplar groves were the land use types that significantly supplied most gas regulation (159.36, 138, and 36 CO2 eq.t/ha · year, respectively), the different provision among each one of them also being significant ($P < 0.001$). Both dry and irrigated cereal crops did not provide this service over the year, and the supply by urban areas was considered null (see Section “Ecosystem services assessment”).

Soil formation: all land use types except urban areas supplied this service. Unexpectedly, not any significant difference among land use types was found, suggesting that the indicator used was not powerful enough in this area.

Nutrient regulation: all land use types except urban areas supplied this service. Riparian forests provided significantly much higher rates (11%) than any other land use type ($P < 0.01$).

Habitat provision: all land use types provided this service, but riparian forests (RQI scoring = 80.17) were the only type that significantly supplied more than any other land use ($P < 0.05$).

Food provision: fruit groves (9310.83 kg/ha · year), irrigated cereal crops (6125.67 kg/ha · year) and dry cereal crops (2827.33 kg/ha · year) in this order were the only land uses that provided this service, and the values differed significantly among these three land uses ($P < 0.001$).

Raw materials production: fruit groves were the main producers of this service. Significant differences ($P < 0.001$) among the land use types providing raw materials were found (i.e. fruit groves 91.79 t/ha · year, riparian forests 79.87 t/ha · year, and poplar groves 15.36 t/ha · year).

Education: only riparian forests and urban areas supplied this service, but riparian forests supplied significantly the most ($P < 0.05$).

Recreation: riparian forests, fruit groves and poplar groves, and urban areas supplied this service, but no significant differences among them were found, probably due to the low number of sites used as indicators of recreational use found in the study area.

Plant diversity assessment across land use types

In the context of the taxonomic approach, our results showed that abandoned crops were the land use with the highest species richness (SR) while riparian forests hosted the most true species diversity (TSD) and species abundance (SA) (Fig 2b, see averaged values in Table A.1). For instance, SR was significantly higher in abandoned crops than in irrigated cereal crops and poplar groves ($P < 0.005$). TSD and SA were also significantly higher in natural or semi-natural habitats (i.e. riparian forests and abandoned crops) than in cultivated lands (i.e. dry and irrigated cereal crops, fruit groves and poplar groves) ($P < 0.001$). Similarly, the functional approach revealed that growth-forms richness (GR) was significantly lower in dry and irrigated cereal crops than in riparian forests ($P < 0.01$) and fruit groves ($P < 0.05$). Growth-forms abundance (GA) followed the same pattern as the taxonomic approach (i.e. being significantly higher in riparian forests and abandoned crops ($P < 0.001$)); however, differences among land use types in true growth-forms diversity (TGD) were not significant, though riparian forests seemed to host higher TGD (see Table A.3 for detailed significant differences between land uses).

Linkages between ecosystem services and plant diversity across land use types

Most correlations between ecosystem services and plant diversity were positive and significant, and many of them (39.58%) were strong ($|0.7| > r > |0.5|$) or very strong ($r > |0.7|$). All diversity indexes except SR were strongly correlated to at least three ecosystem services, but there were two ecosystem services that did not correlated strongly to any diversity index, namely soil formation and nutrient regulation (Table 3). Habitat provision ($0.80 > r > 0.50$) and education ($0.83 > r > 0.50$) positively correlated to most diversity indexes (i.e. each service correlated to 5 diversity indexes), followed by recreation (correlated to 4 indexes; $0.77 > r > 0.55$), gas regulation (correlated to 2 indexes; $0.68 > r > 0.53$) and raw materials production

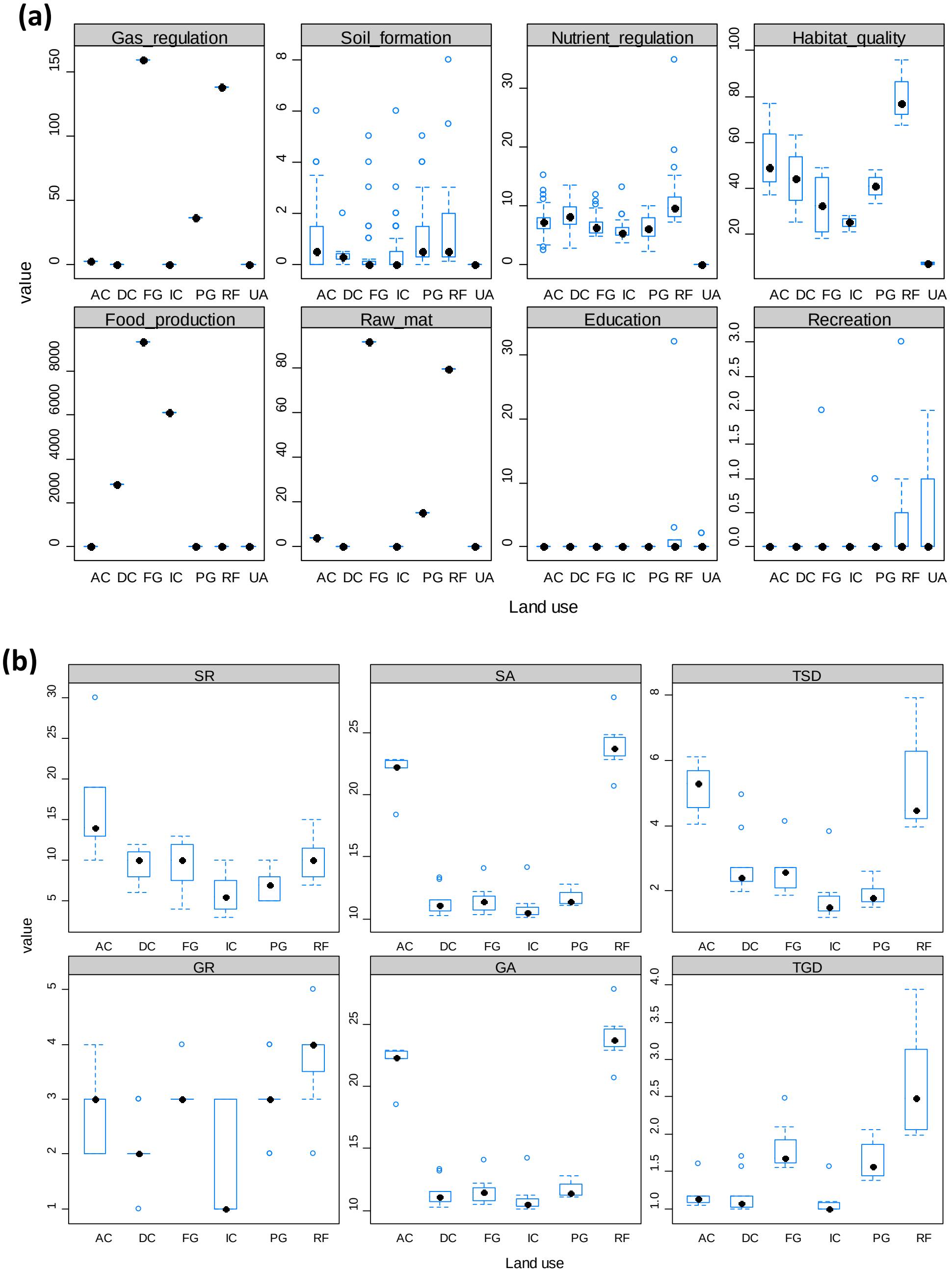

Figure 2. Selected ecosystem services (a) and plant diversity (b) measured at each land use type. Black dots represent the median, boxes comprise 50% of the measurements, dashed lines mark variability outside Q1 and Q3, and white dots are outliers. Units of ecosystem services are provided in Table 2. Plant diversity indexes: SR, Species richness; SA, Species abundance; TSD, True species diversity; GR, Growth-forms richness; GA, Growth-forms abundance; TGD, True growth-forms diversity. Land use abbreviations: AC, Abandoned crops; DC, Dry cereal crops; FG, Fruit groves; IC, Irrigated cereal crops; PG, Poplar groves; RF, Riparian forests; UA, Urban areas.

(correlated to one index; r = 0.57). Food provision was negatively correlated to all diversity indexes, being strongly (r = -0.56) two of them (SA and GA).

Interestingly, analyzing these interactions by land use type resulted in some reversed outcomes (Table 4). For instance, in riparian forests soil formation was very strongly and negatively correlated to SA (r = -0.66) and GA (r = -0.67), and in abandoned crops it was strongly and negatively correlated to TGD (r = -0.51). Besides, in abandoned crops nutrient regulation was very strongly but positively correlated to SA and GA (r = 0.57), and in dry cereal crops strongly correlated to SR (r = 0.54). Additionally, these analyses highlighted the diverse interactions each particular land use type displays between habitat provision and the different components of plant diversity measured. For instance, in poplar groves habitat provision was very strongly and negatively correlated to four diversity indexes (i.e. SA, TSD, GA, and TGD; -0.90 > r > -0.93), and to SR in fruit groves (r = -0.91); however, in abandoned crops it was very strongly and positively correlated to SA, TSD, and GA (0.94 > r > 0.84), and strongly correlated to them in dry cereal crops (0.62 > r > 0.50) and in irrigated cereal crops (0.61 > r > 0.58). In riparian forests, habitat provision was very strongly correlated to SR (r = 0.63) and strongly correlated to GR (r = 0.51). Finally, in fruit groves it was very strongly correlated to TGD (r = 0.70).

Discussion

Our study advances the understanding about the opportunities that floodplains dominated by agricultural uses have to provide multiple ecosystem services while conserving their associated biodiversity. As each land use type of the River Piedra floodplain provided different ecosystem services, its potential to obtain mutual benefits can be implemented through compatible management of its multiple land uses. For instance, all land use types of the River Piedra floodplain provided habitat at a certain extent, and excluding urban areas, also provided soil formation and nutrient regulation. Natural and semi-natural land use types (i.e. riparian forests and abandoned crops, respectively) provided most services (including regulating, supporting, and cultural services) while accommodating the greatest values of most plant diversity indexes (namely, SR, SA, TSD, and GA). Fruit groves, which in the study area were not intensively managed, and riparian forests hosted the greatest values of the remaining diversity indexes (GR and TGD). Moreover, the high number of significant positive correlations between the selected ecosystem services and plant diversity (i.e. five diversity indexes were strongly correlated to three or more ecosystem services) confirmed our hypothesis of the synergic linkages between ecosystem services and biodiversity. Such linkages were especially evident for supporting and cultural services. We also found a direct trade-off between

Table 3. Correlation analyses between selected ecosystem services and plant diversity indexes (n = 165; ·P < 0.1; *P < 0.05; **P < 0.01; ***P < 0.001). Strong correlations (|0.7| > r > |0.5|) are marked in bold, and very strong correlations (r > |0.7|) are marked within a box. Plant diversity indexes: SR, Species richness; SA, Species abundance; TSD, True species diversity; GR, Growth-forms richness; GA, Growth-forms abundance; TGD, True growth-forms diversity.

		Plant diversity					
		SR	**SA**	**TSD**	**GR**	**GA**	**TGD**
Ecosystem services	**Gas regulation**	-0.06	0.20*	0.19*	**0.53*** **	0.20*	**0.68*** **
	Soil formation	-0.01	0.28***	0.37***	0.29***	0.28***	0.38***
	Nutrient regulation	0.15 ·	0.40***	0.35***	0.24**	0.40***	0.35***
	Habitat provision	0.14 ·	**0.80*** **	**0.70*** **	**0.50*** **	**0.80*** **	**0.76*** **
	Food provision	-0.26***	**-0.56*** **	-0.46***	-0.22**	**-0.56*** **	-0.19*
	Raw materials	-0.18*	0.09	0.07	0.34***	0.09	**0.57*** **
	Education	-0.02	**0.59*** **	**0.50*** **	**0.54*** **	**0.59*** **	**0.83*** **
	Recreation	0.02	**0.66*** **	**0.55*** **	0.47***	**0.65*** **	**0.77*** **

Table 4. Correlation analyses between selected ecosystem services and plant diversity indexes by land use type (·P < 0.1; *P < 0.05; **P < 0.01; ***P < 0.001). Note that only those ecosystem services displaying variability among sample points within a land use are shown. Strong correlations (|0.7| > r > |0.5|) are marked in bold, and very strong correlations (r > |0.7|) are marked within a box. Plant diversity indexes: SR, Species richness; SA, Species abundance; TSD, True species diversity; GR, Growth-forms richness; GA, Growth-forms abundance; TGD, True growth-forms diversity. Land use abbreviations: AC, Abandoned crops; DC, Dry cereal crops; FG, Fruit groves; IC, Irrigated cereal crops; PG, Poplar groves; RF, Riparian forests.

		Plant diversity					
Ecosystem services	**Land use types**	**SR**	**SA**	**TSD**	**GR**	**GA**	**TGD**
Soil formation	**AC (n = 30)**	-0.46*	0.16	0.01	-0.19	0.15	**-0.51****
	DC (n = 27)	-0.04	0.09	0.03	0.33 ·	0.09	0.22
	FG (n = 28)	0.09	0.13	-0.03	-0.09	0.13	-0.25
	IC (n = 27)	-0.03	0.18	0.18	0.05	0.18	0.08
	PG (n = 27)	0.22	0.02	-0.12	0.27	0.02	-0.08
	RF (n = 26)	0.18	**-0.66*****	0.44*	0.16	**-0.67*****	0.32
Nutrient regulation	**AC (n = 30)**	0.30	**0.57*****	**0.56****	0.27	**0.57*****	0.30
	DC (n = 27)	**0.54****	0.02	0.11	0.21	0.03	0.05
	FG (n = 28)	0.07	-0.21	-0.18	0.20	-0.21	-0.05
	IC (n = 27)	-0.02	-0.13	-0.14	0.09	-0.13	-0.09
	PG (n = 27)	-0.41*	-0.02	0.06	0.02	-0.02	0.18
	RF (n = 26)	-0.11	0.30	-0.07	-0.38 ·	0.30	-0.13
Habitat provision	**AC (n = 30)**	0.27	$\boxed{\mathbf{0.94***}}$	$\boxed{\mathbf{0.84***}}$	0.44*	$\boxed{\mathbf{0.94***}}$	0.26
	DC (n = 27)	-0.38*	**0.62*****	**0.50****	0.14	**0.61*****	0.36 ·
	FG (n = 28)	$\boxed{\mathbf{-0.91***}}$	0.33 ·	0.37 ·	-0.31	0.31	**0.70*****
	IC (n = 27)	-0.21	**0.61*****	**0.58****	0.00	**0.60*****	0.43*
	PG (n = 27)	-0.10	$\boxed{\mathbf{-0.93***}}$	$\boxed{\mathbf{-0.90***}}$	0.00	$\boxed{\mathbf{-0.93***}}$	$\boxed{\mathbf{-0.92***}}$
	RF (n = 26)	**0.63*****	0.48*	0.29	**0.51****	0.49*	0.47*

provisioning services and plant diversity. However, this trade-off might be reduced through compatible land use policies. For instance, the preservation of a forested belt alongside the riverbanks of cultivated lands would enhance biodiversity conservation and multiple services related to riparian habitats (e.g. water and nutrient regulation, soil retention, waste treatment, aquatic species nursery, recreation; Loomis et al., 2000; Sweeney et al., 2004) with insignificant yield loss (Polasky et al., 2005).

The River Piedra floodplain works as a multifunctional mosaic of habitats where each land use type provides different amounts of each service (Felipe-Lucia et al., 2014) due to the different species it comprises, and the different functions these perform in the ecosystem (De Groot et al., 2010; Isbell et al., 2011). Indeed, some ecosystem services were provided uniquely by certain land uses. For instance, education was only provided by riparian forests and urban areas; recreation was provided by fruit and poplar groves, riparian forests, and urban areas; and food provision was only supplied by dry and irrigated cereal crops and fruit groves. Therefore, in the River Piedra floodplain and likely in other floodplain agroecosystems with similar characteristics, increasing the riparian forests surface while maintaining the productive crops would improve both ecosystem services provision and biodiversity conservation (Benayas et al., 2009) at floodplain scale. Moreover, in floodplains extensively used for agriculture, increasing natural and semi-natural habitats through ecological restoration (Srivastava and Vellend, 2005) together with an appropriate management of the cultivated lands (e.g. reduced fertilizers inputs and use of pesticides, keeping edges and buffer zones) is critical given that riparian forests and multifunctional floodplains in Europe have almost disappeared (Tockner et al., 2009). Active restoration is recommended in degraded floodplains with low river dynamics, such as in large agricultural areas, because in these areas vegetation is dominated by degraded communities not representative of riparian forests which provide the most ecosystem services (Felipe-Lucia et al., 2014). Additionally, spontaneous recovery to riparian forest takes a long time, even after restoration (Moreno-Mateos et al. 2012). These management practices would halt biodiversity losses (Billeter et al., 2008), especially acute for Spanish fishes and amphibians (Vidal-Abarca and Suárez, 2013); reduce current disservices caused by agriculture, such as loss of habitat wildlife, nutrient runoff, and sedimentation on water ways (Zhang et al., 2007; Power, 2010); and reverse disservices provided in the past, such as water pollution, health risks, and biodiversity loss (Swinton et al., 2007).

Although we did not compare the same land use types in other landscapes, our results concur with recent studies (e.g. Harrison et al., 2010; Petz and van Oudenhoven, 2012) which found that cultivated lands (i.e. dry and irrigated cereal crops, fruit groves and poplar groves), fundamentally supplied provisioning services whereas natural and semi-natural habitats provided most of the other services but food supply. However, the results of this study can be applied to other floodplains with similar characteristics, such as floodplains with large areas and rivers with a large part of the floodplain area occupied by extensive agricultural uses, which is a common characteristic of floodplains in European and North American countries; but specific land planning should be adapted to each specific context (Mascarenhas et al. 2014). As indicated above, potential trade-offs between provisioning and other services could be solved through adequate land use policies. Currently, the River Piedra floodplain is not managed according to any specific plan to promote multifunctionality. Rather, farmers manage their properties according to their own individual interests. Generally, the main motivation for crop selection is to be eligible to benefit from European subsidies (e.g. the Common Agricultural Policy) as otherwise crops are not cost-effective enough to make a living there. Additionally, the main regulation for land use in Spanish floodplains (i.e. the Public Hydraulic Domain; BOE, 2008) is usually overlooked. This regulation establishes a 100 m of limited use between the river and any private use of the riverbanks and a 5 m buffer of public use, but in most of the River Piedra floodplain it is not implemented. In this area, the lack of commitment to comply with such regulations diminishes the potential of the floodplain to supply ecosystem services. Designing participatory land use policies based on the integration of ecosystem conservation goals as a part of farmers' activities, and including within the agricultural fields a reserve for recovering riparian forest, would reduce the potential trade-offs and improve the provision of ecosystem services by this landscape. This is established as a key point in the new European Common Agricultural Policy (APP Brief No. 5), which based most of the development in rural areas on the so called "greening" activities. Multifunctional landscapes have been argued to supply multiple ecosystem services in other Mediterranean agroecosystems (Willaarts et al. 2012), in semi-arid agroecosystems (O'Farrell et al., 2010), and in forests (Mitchell et al. 2014), by adapting specific land use polices to conserve their associate biodiversity (Gottschalk et al., 2007) and to benefit from the different ecosystem services each land use provides.

In addition, our results support current research highlighting that high diversity is related to high ecosystem services provision (Chan et al., 2006; Díaz et al., 2006; Palumbi et al., 2008; Turner et al., 2007; Bello et al., 2010). By assessing biodiversity using several indexes and from two different approaches (namely, the taxonomic and the functional approach), we have been able to specify which aspects of biodiversity are correlated to each ecosystem service in our study area. For instance, both approaches of abundance and true diversity indexes were associated to cultural services and habitat quality. However, richness indexes showed weaker correlation to ecosystem services and abundance indexes displayed negative relationships with food provision.

We acknowledge that there is a debate about designing policies to support both ecosystem services and biodiversity conservation (e.g. Redford and Adams, 2009; Skroch and López-Hoffman, 2010; Reyers et al. 2012; Faith, 2012), which anticipates likely uneven outcomes (Balvanera et al., 2006; Nelson et al., 2009, 2008). The difficulties to find joint solutions increase when attempting to implement policies at larger scales (Chan et al., 2006; Naidoo et al., 2008), but might decrease if policies are planned and implemented at smaller scales (Turner et al., 2007). Our study addresses the finest spatial scale at which policies can be implemented, which is the piece of land under different uses. In fact, our analyses reveal that the significance in ecosystem services-biodiversity interactions vary across land use types. For instance, habitat provision was strongly correlated to most plant diversity indexes in riparian forests, dry and irrigated cereal crops, and abandoned crops, but in poplar groves it was very strongly and negatively correlated to most indexes. These analyses enable to distinguish the land use type in which linkages occur, highlighting the land use type as an essential controlling factor in such interactions. Additionally, despite most studies relating ecosystem services to biodiversity base their results solely on species richness analyses (Balvanera et al., 2006), our results show that correlations between ecosystem services and biodiversity change according to the indicator used to assess biodiversity. Thus, assessing ecosystem services and biodiversity in each land use separately and considering several measures of biodiversity would enable us to gain a better understanding of the ecosystem. A sound knowledge of the area of interest is crucial to design specific management policies aiming to enhance both ecosystem services and biodiversity conservation together.

Our study also reveals that some indicators were not useful for stressing significant differences in the provision of ecosystem services across the land use types of our study area (e.g. the organic matter layer depth as an indicator of soil formation, highly variable likely due to the effect of the distance to the river; and the number of areas for social amenity as an indicator of recreation, possibly due to the low number of samples to be recorded). Research on selecting and using indicators to evaluate ecosystem services is of high interest as it is for land management (e.g. the project ESID (Ecosystem Service Indicator Database), from the World Resources Institute) (Müller et al., 2006; De Groot et al., 2010; Van Oudenhoven et al., 2012). This type of research expands the potential of ecosystem services science, investigating suitable indicators to discriminate ecosystem service provision across land use types and ecosystems, which may be useful to design specific land management policies. Additionally, a better understanding of the interactions among land use types is required to determine how ecosystem services and biodiversity can be enhanced together (Hooper et al., 2005; Haines-Young, 2009). In spite of being driven by sectorial interests, floodplain agroecosystems should be managed as a whole to ensure the sufficient provision of services together with coherence in biodiversity protection policies. Studies aiming to set a framework for sustainable land management enhancing ecosystem services, biodiversity conservation, and human well-being propose intermediate economic development instead of a higher development (Tallis et al., 2008). In fact, most ecological models showed that intermediate development levels are the most resilient. Recognizing the limits of ecosystems to provide services (Kremen, 2005; Dobson et al., 2006; Raudsepp-Hearne et al., 2010) and defining actual societal requirements will set the foundations for sustainable living, based on a balanced management between societal demands and the ability of ecosystems to provide services (Vidal-Abarca et al., 2014). Moreover, understanding the key services that provide each land use type and prioritizing these services (Harrison et al., 2010) to reach a balanced provision of ecosystem services will enhance the benefits each region can supply to society.

In conclusion, the results of this study suggest that in agricultural floodplains, a mosaic landscape of different land use types helps support ecosystem services and contributes to maintaining biodiversity while using local resources. In turn, our results underpin scientific and institutional efforts in policies to connect biodiversity conservation with ecosystem services enhancement. Such policies might be adaptive, i.e. managing agricultural floodplains at the landscape scale while still being able to accommodate specific measures for each land use type. Moreover, riparian forests should be preserved and restored across the floodplain (Luck et al., 2009; Meli et al., 2014) as they are hot spots for biodiversity and ecosystem services provision.

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Appendix

Table A.1. Averaged values of selected ecosystem services and plant diversity provided by each land use type in the floodplain of the River Piedra. Abbreviations: OM, Organic Matter; RQI, Riparian Quality Index; No., number; sp., species; g-f., growth-forms. Land use types: AC, Abandoned crops; DC, Dry cereal crops; FG, Fruit groves; IC, Irrigated cereal crops; PG, Poplar groves; RF, Riparian forests; UA, Urban areas. Biodiversity indexes: SR, Species richness; SA, Species abundance; TSD, True species diversity; GR, Growth-forms richness; GA, Growth-forms abundance; TGD, True growth-forms diversity. Note: for printing purposes this table has been split into two parts.

		Selected Ecosystem Services							
		Gas regulation	**Soil formation**	**Nutrient regulation**	**Habitat provision**	**Food provision**	**Raw materials**	**Education**	**Recreation**
		CO_2 eq. t/ha · year	cm	OM%	Scoring	kg/ha · year	t/ha · year	Items	Items
Land use	**AC**	2.26	1.08	7.29	47.60	0.00	4.15	0.00	0.00
	DC	0.00	0.31	8.13	44.00	2827.33	0.00	0.00	0.00
	FG	159.36	0.51	6.75	32.75	9310.83	91.79	0.00	0.17
	IC	0.00	0.57	5.79	24.50	6125.67	0.00	0.00	0.00
	PG	36.00	1.00	6.25	40.67	0.00	15.36	0.00	0.08
	RF	138.00	1.30	11.00	80.17	0.00	79.87	3.08	0.42
	UA	0.00	0.00	0.00	7.08	0.00	0.00	0.33	0.42

		Plant diversity					
		SR	**SA**	**TSD**	**GR**	**GA**	**TGD**
		No. of sp.	Sp. abundance	Effective no. of sp.	No. of g-f.	G-f. abundance	Effective no. of g-f.
Land use	**AC**	17.20	21.67	5.14	2.80	21.76	1.21
	DC	9.56	11.40	2.80	2.11	11.41	1.19
	FG	9.43	11.56	2.59	3.14	11.60	1.83
	IC	5.88	10.99	1.80	1.75	11.00	1.09
	PG	6.78	11.67	1.89	3.00	11.67	1.66
	RF	10.14	23.96	5.34	3.71	23.97	2.69

(Tabla A.1 cont)

Table A.2 Multicomparison test showing significant differences among land use types in ecosystem services provision (·P < 0.1; *P < 0.05; **P < 0.01; ***P < 0.001). Note that *soil formation* and *recreation* were excluded because they did not show significant differences among land uses. Land use abbreviations: AC, Abandoned crops; DC, Dry cereal crops; FG, Fruit groves; IC, Irrigated cereal crops; PG, Poplar groves; RF, Riparian forests; UA, Urban areas.

	Selected ecosystem services											
	Gas regulation		Nutrient regulation		Habitat provision		Food provision		Raw materials		Education	
Land uses	Pr(>|t|)	Sign.	Pr(>|z|)	Sign.	Pr(>|t|)	Sign.	Pr(>|t|)	Sign.	Pr(>|t|)	Sign.	Pr(>|z|)	Sign.
DC - AC	<0.001	***	1.00		1.00		<1e-10	***	<2e-16	***	1	
FG - AC	<0.001	***	1.00		0.69		<1e-10	***	<2e-16	***	1	
IC - AC	<0.001	***	0.41		0.32		<1e-10	***	<2e-16	***	1	
PG - AC	<0.001	***	0.99		0.99		1		<2e-16	***	1	
RF - AC	<0.001	***	0.00	**	0.07	.	1		<2e-16	***	0.04	*
UA - AC	<0.001	***	<0.001	***	0.01	*	1		<2e-16	***	0.94	
FG - DC	<0.001	***	0.99		0.93		<1e-10	***	<2e-16	***	1	
IC - DC	0.11		0.31		0.63		<1e-10	***	1		1	
PG - DC	<0.001	***	0.95		1.00		<1e-10	***	<2e-16	***	1	
RF - DC	<0.001	***	0.04	*	0.07	.	<1e-10	***	<2e-16	***	0.04	*
UA - DC	0.05	*	<0.001	***	0.06	.	<1e-10	***	1		0.94	
IC - FG	<0.001	***	0.76		0.99		<1e-10	***	<2e-16	***	1	
PG - FG	<0.001	***	1.00		0.99		<1e-10	***	<2e-16	***	1	
RF - FG	<0.001	***	0.00	**	0.01	**	<1e-10	***	<2e-16	***	0.04	*
UA - FG	<0.001	***	<0.001	***	0.26		<1e-10	***	<2e-16	***	0.94	
PG - IC	<0.001	***	0.89		0.79		<1e-10	***	<2e-16	***	1	
RF - IC	<0.001	***	<0.001	***	0.00	**	<1e-10	***	<2e-16	***	0.04	*
UA - IC	1.00		<0.001	***	0.72		<1e-10	***	1		0.94	
RF - PG	<0.001	***	<0.001	***	0.04	*	1		<2e-16	***	0.04	*
UA - PG	<0.001	***	<0.001	***	0.10		1		<2e-16	***	0.94	
UA - RF	<0.001	***	<0.001	***	<0.001	***	1		<2e-16	***	0.44	

Table A.3 Multicomparison test showing significant differences among land use types in plant diversity (·$P < 0.1$; *$P < 0.05$; **$P < 0.01$; ***$P < 0.001$). Note that *TGD* was excluded because there was no significant difference among land uses. Land use abbreviations: AC, Abandoned crops; DC, Dry cereal crops; FG, Fruit groves; IC, Irrigated cereal crops; PG, Poplar groves; RF, Riparian forests. Plant diversity indexes: SR, Species richness; SA, Species abundance; TSD, True species diversity; GR, Growth-forms richness; GA, Growth-forms abundance.

<table>
<tr><th></th><th colspan="10">Plant diversity</th></tr>
<tr><th></th><th colspan="2">SR</th><th colspan="2">SA</th><th colspan="2">TSD</th><th colspan="2">GR</th><th colspan="2">GA</th></tr>
<tr><th>Land uses</th><th>Pr(>|z|)</th><th>Sign.</th><th>Pr(>|z|)</th><th>Sign.</th><th>Pr(>|z|)</th><th>Sign.</th><th>Pr(>|z|)</th><th>Sign.</th><th>Pr(>|z|)</th><th>Sign.</th></tr>
<tr><td>DC - AC</td><td>0.105</td><td></td><td><0.001</td><td>***</td><td>0.002</td><td>**</td><td>0.716</td><td></td><td><0.001</td><td>***</td></tr>
<tr><td>FG - AC</td><td>0.103</td><td></td><td><0.001</td><td>***</td><td>0.001</td><td>***</td><td>0.992</td><td></td><td><0.001</td><td>***</td></tr>
<tr><td>IC - AC</td><td><0.001</td><td>***</td><td><0.001</td><td>***</td><td><0.0001</td><td>***</td><td>0.255</td><td></td><td><0.001</td><td>***</td></tr>
<tr><td>PG - AC</td><td>0.002</td><td>**</td><td><0.001</td><td>***</td><td><0.0001</td><td>***</td><td>0.999</td><td></td><td><0.001</td><td>***</td></tr>
<tr><td>RF - AC</td><td>0.213</td><td></td><td>0.061</td><td>.</td><td>0.999</td><td></td><td>0.526</td><td></td><td>0.078</td><td>.</td></tr>
<tr><td>FG - DC</td><td>1</td><td></td><td>0.999</td><td></td><td>0.999</td><td></td><td>0.257</td><td></td><td>0.999</td><td></td></tr>
<tr><td>IC - DC</td><td>0.474</td><td></td><td>0.991</td><td></td><td>0.557</td><td></td><td>0.956</td><td></td><td>0.990</td><td></td></tr>
<tr><td>PG - DC</td><td>0.824</td><td></td><td>0.998</td><td></td><td>0.663</td><td></td><td>0.359</td><td></td><td>0.998</td><td></td></tr>
<tr><td>RF - DC</td><td>0.999</td><td></td><td><0.001</td><td>***</td><td>0.0003</td><td>***</td><td>0.008</td><td>**</td><td><0.001</td><td>***</td></tr>
<tr><td>IC - FG</td><td>0.548</td><td></td><td>0.954</td><td></td><td>0.806</td><td></td><td>0.039</td><td>*</td><td>0.939</td><td></td></tr>
<tr><td>PG - FG</td><td>0.870</td><td></td><td>1.000</td><td></td><td>0.881</td><td></td><td>0.999</td><td></td><td>1.000</td><td></td></tr>
<tr><td>RF - FG</td><td>0.999</td><td></td><td><0.001</td><td>***</td><td>0.0001</td><td>***</td><td>0.835</td><td></td><td><0.001</td><td>***</td></tr>
<tr><td>PG - IC</td><td>0.993</td><td></td><td>0.913</td><td></td><td>0.999</td><td></td><td>0.059</td><td>.</td><td>0.913</td><td></td></tr>
<tr><td>RF - IC</td><td>0.335</td><td></td><td><0.001</td><td>***</td><td><0.0001</td><td>***</td><td><0.001</td><td>***</td><td><0.001</td><td>***</td></tr>
<tr><td>RF - PG</td><td>0.686</td><td></td><td><0.001</td><td>***</td><td><0.0001</td><td>***</td><td>0.636</td><td></td><td><0.001</td><td>***</td></tr>
</table>

CAPÍTULO 4. INTERACTIONS AMONG ECOSYSTEM SERVICES ACROSS LAND USES IN A FLOODPLAIN AGROECOSYSTEM*

ABSTRACT. Managing human-dominated landscapes such as agroecosystems is one of the main challenges facing society today. Decisions about land-use management in agroecosystems involve spatial and temporal trade-offs. The key scales at which these trades-offs occur are poorly understood for most systems, and quantitative assessments of the services provided by agroecosystems under different combinations of land uses are rare. To fill these knowledge gaps, we measured 12 ecosystem services (ES), including climate regulation, gas regulation, soil stability, nutrient regulation, habitat quality, raw material production, food production, fishing, sports, recreation, education, and social relationships, in seven common land-use types at three spatial scales, i.e., patch, municipality, and landscape, in a riparian floodplain in Spain. We identified the provision of each ES in each land-use type either by direct measurement or from public databases. We analysed the interactions, i.e., trade-offs and synergies, among ES across land uses and spatial scales and estimated ES provision in several land-use change scenarios. Our results illustrated that each land-use type provides unique bundles of ES and that the spatial scale at which measurements were taken affected the mixture of services. For instance, a land-use type with low provision of services per hectare but with an extensive area can supply more services to the overall landscape than a land-use type supplying higher values of services per hectare but with a smaller extent. Hence, riparian forest supplied the most service of any land use type at the patch scale, but dry cereal croplands provided the most services across the municipality and landscape because of their large area. We found that most ES should be managed primarily at the patch scale, but food production, fishing, and social relationships were more relevant to manage at the municipality scale. There was great variability in ES interactions across scales with different causes of trade-offs at each scale. We identified more significant synergies among ES than trade-offs. Trade-offs were originated because some services were mutually incompatible within a given land use, whereas the provision of others depended on land-management decisions within a land-use type. Thus, we propose a classification of ES interactions that incorporates societal values as drivers of management decisions along with biophysical factors as likely causes of ES trade-offs and conclude with practical suggestions to reduce trade-offs and to enhance the supply of multiple ES to society.

Key Words: agroecosystem; ecosystem services; floodplain; interactions; land uses; spatial scales; trade-offs

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Introduction

Agroecosystems are managed to fulfill basic human needs, such as food and raw materials (Zhang et al. 2007). They occupy 40% of the global terrestrial surface (FAO 2009, as cited in Power 2010), of which 3.5% are on floodplains (Tockner and Standford 2002). Floodplains sustain a large portion of the world's food production thanks to their nutrient-rich and water abundant soils. Indeed, great parts of floodplains' extent are dedicated to agricultural production, from 11% in African rivers to 79% in European rivers (Tockner and Standford 2002). Current pressure on floodplain agroecosystems to feed the growing human population is leading to major environmental degradation, including deforestation, soil erosion, nutrient leaching and water abstraction, diversion, and pollution (Simoncini 2009). This is especially important given that floodplains are one of the most endangered habitats and biodiversity hotspots while still the second highest worldwide attraction for housing developers (Moss and Monstadt 2008). Floodplains are key ecosystems for land managers because of their important role in food supply, human settlement, and biodiversity conservation.

Decisions about land-use planning, in floodplains and elsewhere, generally involve spatial and temporal trade-offs for society and ecosystems (Box 1). Consequences of these trade-offs need to be assessed across temporal and spatial scales by policy makers prior to management actions such that managers can make effective decisions (Rodríguez et al. 2005, Tallis and Polasky 2009, Cabell and Oelofse 2012). Such assessments are paramount to maximizing human well-being, enabling adaptive management, and improving resilience in the social-ecological system (Carpenter et al. 2005). The spatial patterns of social-ecological systems, e.g., the number, location, and relative proportion of different land-use types, can vary at differing spatial scales, which can then influence ecological functions (Pringle et al. 2010). Repercussions of outcomes at a particular spatial scale may affect biodiversity and ecosystem conservation, as well as stakeholder interests and institutional responsibilities (Hein et al. 2006). Thus, to make effective land-management decisions, baseline data about the biophysical and social settings are required at the spatial scales of the decisions being made (DeFries et al. 2004, Nicholson et al. 2009). Effects of management actions may have different results across spatial scales (Concepción et al. 2012), e.g., at the individual patch level compared to a municipality, or entire landscape. Therefore high quality local data and multiscale analyses are fundamental to design adequate management plans, understand the trade-offs they encompass, and facilitate decision making (Carpenter et al. 2009).

To orient decision-makers to identifying trade-offs and synergies in land-use planning many studies have applied the concept of ecosystem services (ES; Costanza et al. 1997, de Groot et al. 2002), i.e., the benefits humans obtain from ecosystems, such as clean air, water, food, raw materials, and recreation (Rose and Chapman 2003, Bennett and Balvanera 2007, Barral and Maceira 2012, Rathwell and Peterson 2012). The amount of each ES supplied in a given area depends on both the per hectare provision of service by land-use type and the total amount of each land use found in the study area. Moreover, ES do not operate independently from each other (Pereira et al. 2005), but they interact in trade-offs and synergies. There is also evidence that ES act differently across both spatial and temporal scales (Swift et al. 2004, Rodríguez et al. 2005, Power 2010), and that land-use patterns affect ES provision (Mitchell et al. 2013); however, the key scales for ES management still remain poorly understood (Hein et al. 2006). Most studies of multiple ES use GIS and satellite images (Kreuter et al. 2001, Konarska et al. 2002, Chen et al. 2009), global databases (Viglizzo and Frank 2006, Tianhong et al. 2010), or models to estimate ES provision (Nelson et al. 2009, Goldstein et al. 2012). Few studies however, have gathered local data across land uses (but see Raudsepp-Hearne et al. 2010), despite evidence that these data are critical to accurate assessment of service provision (Eigenbrod et al. 2010)

Floodplains contribute to the provision of more than 25% of all terrestrial ES (Tockner and Standford 2002). Therefore, understanding ES interactions in floodplain agroecosystems is an important challenge in ecology. Understanding how floodplains can be managed across spatial scales to deliver multiple ES could enhance the supply of ES to society, providing land managers with decision-making tools to reach win-win or small loss–big gain solutions (DeFries et al. 2004) for policy-making.

Box 1. Definitions applied to ecosystem services.

Trade-off: Situation in which land use or management actions increase the provision of one ecosystem service and decrease the provision of another. This may be caused by simultaneous responses to the same driver or caused by true interactions among services (Bennett et al. 2009).

Synergy: Situation in which the combined effect of a number of drivers acting on ecosystem services is greater than the sum of their separate effects (adapted from Carpenter et al. 2005). In other words, a synergism occurs when ecosystem services interact with one another in a multiplicative or exponential fashion (Rodríguez et al. 2006). These can be positive, i.e., multiple services improving in provision, or negative, i.e., multiple services declining in provision.

We aim to identify: (1) how the supply of a set of ES changes across land-use types and spatial scales in a floodplain; (2) which trade-offs and synergies are common or different in each land-use type and across spatial scales; and (3) how land-use change might affect the provision of ES. We evaluated 12 ES in 7 land-use types identified within a river floodplain at multiple spatial scales, i.e., patch, municipality and landscape, using ecological indicators. We illustrate significant differences in the supply of ES across land uses, spatial scales and alternative scenarios; and we analyze their interactions, i.e., trade-offs and synergies. Finally, we discuss major questions on ES interactions and suggest practical land-use management applications.

Methods

Study site

The study area is the floodplain of the Piedra River in central Spain (Fig. 1). The Piedra River is 76 km long and the watershed covers an area of 922.72 km^2. The river floodplain ranges from 50 to 300 m wide and occupies 19.3 km^2. It is composed of 12 municipalities in which 1539 people live permanently (Table 1), although the population can double during the summer (Felipe-Lucia 2012). The floodplain is commonly split into three parts, i.e., upper, central, and downstream, based on the amount of water available for agricultural use. In the upper floodplain, the river is dry for most of the year and dry cereal crops are cultivated. The central floodplain, which is rich in water springs, is devoted to highly productive irrigated cereal crops and poplar groves. The downstream floodplain, separated from the central floodplain by a reservoir, is characterized by orchards, fruit groves and abandoned agricultural lands. There are also two main natural areas in the watershed. One of them is located in the upper floodplain gorges and the other, just upstream from the reservoir, is a private natural park whose waterfalls attract thousands of tourists each year.

Data gathering and analyses

We selected 12 ES to measure based on their importance for the ecological functioning of a river floodplain (see Harrison et al. 2010) and our ability to assess them in the study area (Table 2). We estimated the provision of these ES in seven different land use types common to the Piedra River floodplain (Table 3). We measured the area of each land-use type using the latest Spanish crop and land-use digital map (Ministerio de Medio Ambiente Y Medio Rural y Marino 2009) with ArcGIS 10 (ESRI 2012) and validated these measurements with field observations. We assessed ES at three spatial scales, i.e., patch, municipality, and landscape, in which municipality scale comprises the portion of each municipality within the river

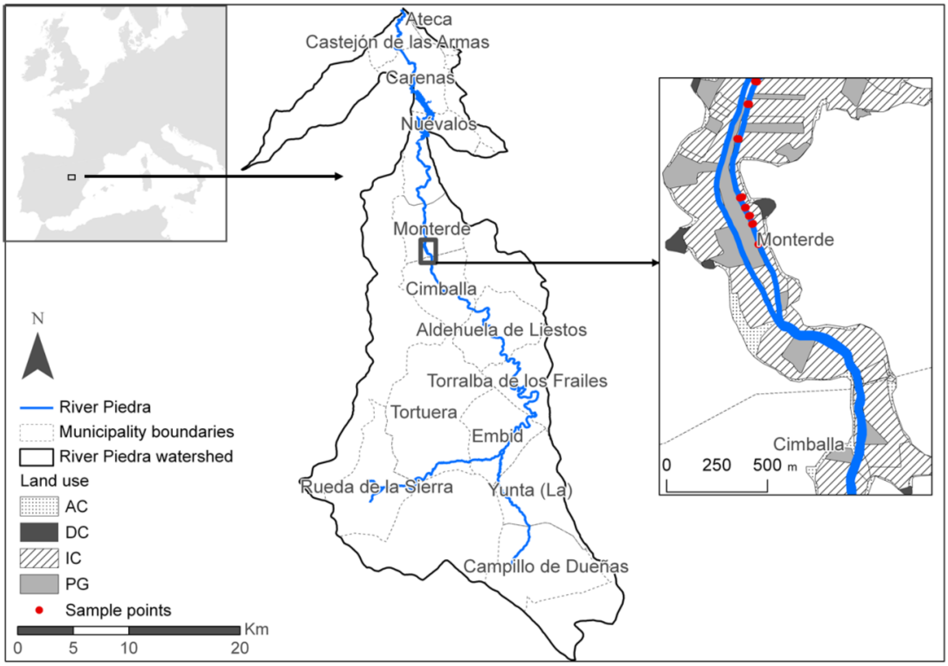

Fig. 1. Location of the study area in Spain (left). Piedra River watershed and municipalities traversed by the Piedra River (center). On the right, detail of the river floodplain spatial distribution showing some land-use types and sample points.

Table 1. Municipalities, number of inhabitants, and extent of each municipality within the floodplain of the Piedra River valley.

Municipality	Number of inhabitants	Total area (km^2)	Area within the floodplain (km^2)	% of land within the floodplain
Aldehuela de Liestos	52	38.04	1.13	2.97
Campillo de Dueñas	92	60.63	0.56	0.93
Carenas	195	31.22	2.87	9.19
Castejón de las Armas	121	16.19	0.78	4.84
Cimballa	127	31.95	1.51	4.72
Embid	53	36.2	2.09	5.79
Monterde	190	55.94	2.06	3.68
Nuévalos	355	41.84	3.39	8.11
Rueda de la Sierra	51	51.02	1.67	3.28
Torralba de los Frailes	94	59.22	0.82	1.38
Torrubia	24	28.18	1.38	4.91
Tortuera	185	82.21	1.06	1.29
Total	**1539**	**532.64**	**19.33**	**3.63**

Modified from Instituto Aragonés de Estadística (Padron Municipal of Inhabitants), updated on 1 January 2011 and from Instituto Nacional de Estadística (Municipal surface), updated on 1 January 2008.

floodplain, ranging from 0.5 to 3.4 km^2 in area and including several land-use types, and landscape scale refers to the entire floodplain of the Piedra River catchment.

To assess ES provision, we either estimated the ES indicators directly or obtained the values for ES indicators from public databases (Table 2, see Appendix 1 for details). For directly sampled ES, we surveyed three 0.5 ha patches in representative sites of each land-use type distributed throughout the river floodplain. Within each of these patches, three floodplain-wide transects perpendicular to the river channel were established 25 m apart. The appropriate indicators were measured at 1 m, 5 m, and 15 m away from the river along each transect. These values were averaged to determine the overall mean provision of services in that land-use type at the patch scale. Data obtained from databases were available as average values per hectare by land-use type, except for the services of fishing areas and sports, which were available as mapped trails and their length was measured using GIS tools.

For regulating, supporting, and provisioning services, we first quantified ES provided within individual patches of unique land use, and used this data to estimate total provision at the municipality and landscape scales based on the total area of each land-use type at each of these scales. Thus, to scale from the patch to municipality, ES values at the patch scale were multiplied by the extent of each land-use type within each municipality. Average values of each ES by land-use type across all municipalities were used at the municipality scale. To scale up to the landscape, ES values at the patch scale were multiplied by the total extent of each land-use type in the whole river floodplain.

Cultural services were measured at the municipality scale, rather than the patch, and therefore cultural ES were downscaled from the municipality scale to the patch scale by dividing the ES value per municipality by the extent of each land use within each municipality, and averaging. To scale up cultural ES to the landscape, ES values at the municipality scale were aggregated by land-use type (further information about spatial scale adaptation is provided in Appendix 1).

To determine the key spatial scale to manage land-use planning based on the provision of ES, we compared the amount of each service provided relative to other services across the patch, municipality and landscape scales. To do this, we estimated the proportion of each land-use type at each spatial scale and multiplied it by the ES provision values of each land-use type at its corresponding scale (Table 3). We expected this scaling technique to be useful to discriminate the provision of ES by a range of land-use types at different spatial scales because land use extension is independent from municipalities and the landscape. Finally, to simulate scenarios in which a single land use occupied the entire floodplain landscape, we multiplied the ES supply per hectare of each land-use type by the extent of the floodplain landscape.

Table 2. Selected ecosystem services, abbreviations used in following tables and figures, indicators used for their evaluation, units in which they were measured, and data source (See Appendix 1 for detailed information on the data source, indicators used, and units).

ES Group	Ecosystem Service	ES abbrev.	Indicator	Unit	Data source
Regulating	Climate regulation	Climate	Inverse Daily Temperatures Range	$°C^{-1}$	Sampled
Regulating	Gas regulation	Gas	Carbon sequestration by plants	CO_2eqTons /Ha	Database
Regulating	Soil stability	Soil	Organic matter layer in top soil	cm	Sampled
Regulating	Nutrient regulation	Nutrient	Total Nitrogen in top soil	ppm	Sampled
Supporting	Habitat quality	Habitat	Riparian Quality Index	Score	Sampled
Provisioning	Raw materials	Raw_mat	Annual biomass increase	Tons/Ha	Database
Provisioning	Food production	Food	Nutritive productivity	Kcal/Ha	Database
Cultural	Fishing	Fishing	Kilometric Abundance Index	Km/Km	Database
Cultural	Sports	Sports	Trails with a view over the area	Ha	Database
Cultural	Recreation	Recreation	Areas for local amenity	Items	Sampled
Cultural	Education	Education	Notice boards with information about the ecosystem	Items	Sampled
Cultural	Social relationships	Assoc	Nature user local associations (both farmer unions and conservationists)	Number	Sampled

We plotted these results using the graphics package (Murrell 2005) of the statistical software R (R Development Core Team 2012). To detect significant differences in the provision of ES among the studied land uses and spatial scales we performed generalized linear mixed models (GLMM) fitted with the Poisson family distribution using the 'lme4' R package (Bates et al. 2012). For this, we estimated each ES (response variable) as a function of the interaction between land-use type and spatial scale (categorical variables). Models were validated by checking the model residual plots (Zuur et al. 2009). We performed multiple comparison tests ('multcomp' R package; Hothorn et al. 2008) and figures plots ('effects' R package; Fox 2003) to determine significant differences among the means. Finally, to test the significance of ES interactions and their directions, i.e., positive or negative, Spearman correlation from the ‘AED’ R package (Zuur et al. 2009) was applied. Interactions were considered significant positives, but not necessarily synergies, when $r^2 > 0.5$ and significant negatives, and thus, trade-offs, when $r^2 < -0.5$. We also considered the interactions among ES by each land-use type separately using the same techniques.

Table 3. Main land uses identified in the Piedra River floodplain, abbreviations utilized in following tables and figures, proportion of each land-use type at each spatial scale, total extent they occupy, and percentage of the floodplain each one represents. Note that Water includes both the Piedra River and its reservoir of 5.60 km^2 surface and Others refers to minor land uses representing less than 1% each one (e.g. vineyards, almond trees).

Main land uses	Abbrev.	Proportion at patch scale	Proportion at municipality scale	Proportion at landscape scale	Total extent (km^2)	Percentage in the landscape
Abandoned Crops	AC	0.28	0.01	0.02	0.28	1.45
Dry cereal Crops	DC	0.16	0.36	0.38	5.38	27.82
Fruit Groves	FG	0.06	0.08	0.08	1.12	5.81
Irrigated Crops	IC	0.07	0.19	0.13	1.8	9.30
Poplar Groves	PG	0.09	0.04	0.05	0.71	3.66
Riparian Forests	RF	0.20	0.05	0.04	0.51	2.64
Urban areas	UA	0.06	0.06	0.09	1.23	6.34
Others		0.08	0.22	0.22	1.2	6.23
Water					7.1	36.74
Total study area					**19.33**	**100**

Results

Changes in the supply of ecosystem services across land-use types and spatial scales

Each land-use type in our study provided unique mixtures and quantities of ES but some land uses did not provide some ES, for example, urban areas did not supply nutrient regulation. We also noticed that the importance of each land-use type in supplying ES varied across the spatial scales studied (Table 4; see also Appendix 2, Fig. S1). For instance, at the patch scale, riparian forest supplied more soil stability, nutrient regulation, habitat quality, sports, recreation, and education than any other

Table 4. Ecosystem services delivered by different land uses at three spatial scales, i.e., patch, municipality and landscape. (Abbreviations: AC = abandoned crops; DC = dry cereal crops; FG = fruit groves; IC = irrigated cereal crops; PG = poplar groves; RF = riparian forest; UA = urban areas).

	Climate regulation	Gas regulation	Soil stability	Nutrient regulation	Habitat quality	Raw materials	Food production	Fishing	Sports	Recreation	Education	Social relationships
Scale												
Land use	ºC^{-1}	$C0_2$eqTons/Ha	Cm	Ppm	Score	Tons/Ha	Kcal/Ha	m	Ha	Items	Items	Number
Patch												
AC	0.05	2.26	1.08	0.24	47.60	4.15	0	19.45	1.47	0.00	0.00	0.12
DC	0.05	0.00	0.31	0.22	46.25	2.85	9989680	0.78	4.87	0.00	0.00	1.15
FG	0.06	159.36	0.51	0.20	37.94	91.79	4503021	0.02	0.18	0.02	0.00	0.97
IC	0.05	0.00	0.47	0.13	38.35	42.39	23645073	39.10	3.50	0.00	0.00	1.11
PG	0.06	36.00	0.80	0.15	40.67	15.36	0	4.11	1.42	0.01	0.00	0.44
RF	0.05	138.00	1.30	0.34	80.17	79.87	0	16.92	18.58	0.10	0.72	0.26
UA	0.06	0.00	0.00	0.00	0.00	0.00	0	2.78	1.25	0.04	0.03	0.23
Municipality												
AC	0.38	15.87	7.56	1.66	334.27	29.12	0	3641.23	23.31	0.00	0.00	0.12
DC	2.62	0.00	15.15	10.67	2260.74	139.09	488304640	593.16	22.60	0.00	0.00	1.15
FG	1.21	2980.56	9.52	3.80	709.56	1716.75	84221510	13.41	0.39	0.33	0.00	0.97
IC	1.18	0.00	10.49	2.96	861.87	952.77	531393467	5929.18	35.03	0.00	0.00	1.11
PG	1.12	636.30	14.17	2.72	718.78	271.48	0	778.91	12.03	0.25	0.00	0.44
RF	0.26	641.20	6.05	1.58	372.48	371.10	0	2566.12	46.44	0.45	3.36	0.26
UA	0.67	0.00	0.00	0.00	0.00	0.00	0	420.86	2.62	0.45	0.36	0.23
Landscape												
AC	1.51	63.48	30.23	6.63	1337.08	116.49	0	14750.16	93.25	0.00	0.00	0.48
DC	28.83	0.00	166.68	117.40	24868.16	1529.96	5371351039	593.16	248.60	0.00	0.00	12.64
FG	7.28	17883.38	57.10	22.78	4257.35	10300.50	505329062	13.41	2.34	2.00	0.00	5.83
IC	9.46	0.00	83.90	23.66	6894.95	7622.15	4251147735	29645.89	280.25	0.00	0.00	8.87
PG	4.49	2545.20	56.67	10.86	2875.13	1085.92	0	3115.66	48.11	1.00	0.00	1.74
RF	2.81	7053.18	66.60	17.37	4097.32	4082.13	0	12830.61	510.87	5.00	37.00	2.87
UA	7.39	0.00	0.00	0.00	0.00	0.00	0	2104.29	28.86	5.00	4.00	2.57

land-use type. Similarly, fruit groves supplied more climate and gas regulation and raw materials. However, at the municipality and landscape scales, the key service suppliers changed, primarily because of the amount of land in each land-use type. So, for example, dry cereal cropland supplied the most climate regulation, soil stability, nutrient regulation, habitat quality, food production and social relationships across the whole landscape, whereas fruit groves were the main supplier of gas regulation. Riparian forest also supplied the most sports, education and recreation at the municipality and landscape scales.

Across all three spatial scales, three land uses consistently supplied ES in larger amounts than other land uses. They were riparian forests, i.e., ES provided largely in riparian forests were sports, recreation and education; fruit groves, i.e., gas regulation and raw materials; and dry cereal crops, i.e. social relationships. These land uses remain important across scales because they either supply an elevated level of service or cover a fairly extensive area. Further information about the supply of ES by each land use across spatial scales is depicted in Appendix 2 (Fig. S1).

In addition, the comparison among spatial scales about their relative provision of ES showed larger values for most ES at the patch scale, suggesting this is the key scale to manage ES in our study area. However, values of food production, fishing, and social relationships were larger at the municipality scale (Fig. 2; see also Appendix 2, Fig. S2, Tables S1 - S3).

Simulated scenarios

Simulating scenarios in which a single land use occupied the entire floodplain landscape of the study area resulted in a large range of variation in ES provision. We observed that a landscape composed completely of the riparian forest would increase the widest variety of ES, namely, habitat quality, nutrient regulation, soil stability and the majority of cultural services; a landscape specialized in fruit groves would have high levels of gas and climate regulation and raw materials production; a landscape covered by irrigated cereal crops would maximize food production; and a landscape dedicated to dry cereal crops would enhance social relationships (Fig. 3). Because of the differences in services provided across different land-use types, ultimately, preserving a mixture of land-use types is critical to providing a mixture of services in the landscape.

Ecosystem services interactions: trade-offs and synergies across spatial scales and land-use types

Relationships between ES varied across spatial scales. That is, some interactions, as measured by correlation, between ES were significant only at a single scale whereas others were significant at multiple spatial scales. Across scales 21% of ES interactions varied in significance and 19% reversed from positive to negative or vice versa. The significant interactions between ES were 96% positive and just 4% negative (Table 5). However, only four of the significant positive interactions were consistent across all three spatial scales: the synergies among soil stability, nutrient regulation and habitat quality, and the synergy between recreation and education. Significant negative interactions were only observed between climate regulation and two other services, fishing, and sports, at the patch scale. Finally, we found the largest number of significant positive interactions between ES at the scale of the municipality, especially between cultural and provisioning services.

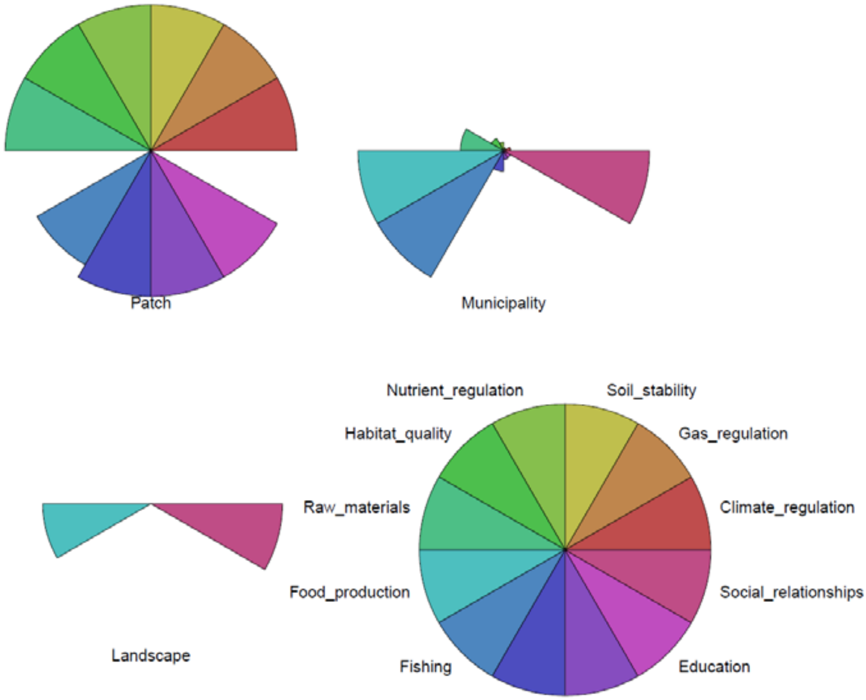

Figure 2. Ecosystem services (ES) supply per hectare across spatial scales. Note that the majority of ES are delivered at patch scale, and only food production, fishing, and social relationships areas are supplied mainly at municipality scale. Pie size represents the relative value in relation to the reference pie chart (the maximum value of the data). Note that empty slices represent the minimum relative value to the contribution of that particular ES.

Including the land-use type as a factor in ES interactions revealed that the only significant correlations were between cultural services and they were all positive. Surprisingly, urban areas were the only land use in which all cultural services correlated among them. In riparian forests fishing, recreation, education, and social relationships were also correlated. Finally, fishing and sports were correlated in all land uses except in riparian forest and abandoned crops (Table 6).

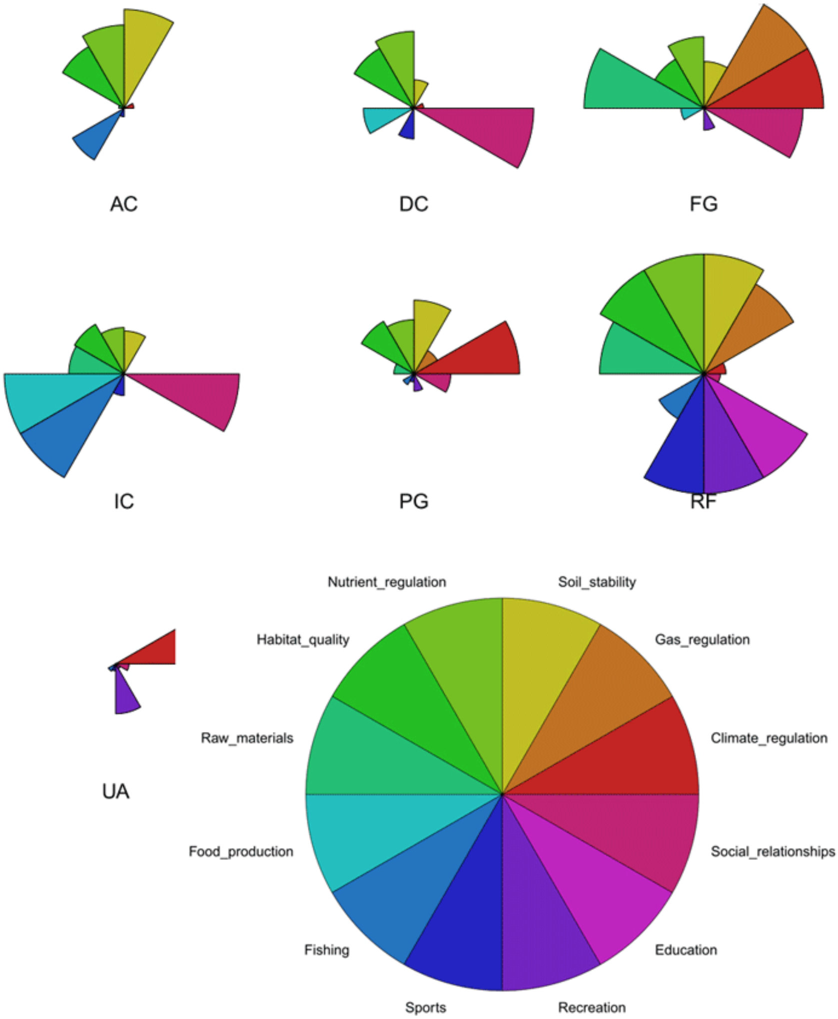

Figure 3. Ecosystem services (ES) scenarios analysis in which we compare the maximum supply of ES by each land use assuming each one occupies the whole floodplain landscape. Pie size represents the relative value in relation to the reference pie chart (the maximum value of the data). Note that empty slices represent the minimum relative value to the contribution of that particular ES. (Abbreviations: AC = abandoned crops; DC = dry cereal crops; FG = fruit groves; IC = irrigated cereal crops; PG = poplar groves; RF = riparian forest; UA = urban areas).

Table 5. Significant interactions ($r^2 > 0.5$ or $r^2 < -0.5$) found between ecosystem services (ES) at three different spatial scales. Note that positive interactions (+) are common and negative interactions (-) are rare. (Abbreviations: p = patch; m = municipality; l = landscape).

	ES																																
	Gas			**Soil**			**Nutrient**			**Habitat**			**Raw_mat**			**Food**			**Fishing**			**Sports**			**Recreation**			**Education**			**Assoc**		
Scale	**p**	**m**	**l**	**p**	**m**	**l**	**p**	**m**	**l**	**p**	**m**	**l**	**p**	**m**	**l**	**p**	**m**	**l**	**p**	**m**	**l**	**p**	**m**	**l**	**p**	**m**	**l**	**p**	**m**	**l**	**p**	**m**	**l**
Climate	+				+			+			+						+	+	-			-	+										+
Gas				+									+	+						+			+			+			+				
Soil							+	+	+	+	+	+		+			+	+					+			+			+				+
Nutrient										+	+	+		+			+	+															+
Habitat																	+	+				+											+
Raw_mat																				+			+			+			+				
Food																															+		+
Fishing																						+	+			+			+				
Sports																										+			+				
Recreation																												+	+	+			
Education																																	

Table 6. Interactions between ecosystem services (ES) across land uses. Note that only significant interactions are shown. All of them were positive, and they were only found among cultural services. (Abbreviations: AC = abandoned crops; DC = dry cereal crops; FG = fruit groves; IC = irrigated cereal crops; PG = poplar groves; RF = riparian forest; UA = urban areas).

	ES																											
	Fishing							Sports							Recreation							Education						
ES	AC	DC	FG	IC	PG	RF	UA	AC	DC	FG	IC	PG	RF	UA	AC	DC	FG	IC	PG	RF	UA	AC	DC	FG	IC	PG	RF	UA
Sports		+	+	+	+		+																					
Recreation			+		+	+	+							+														
Education						+	+							+						+	+							
Assoc			+			+	+							+						+	+						+	+

Discussion

The results of our study support regional-level studies in finding that the supply of ES varies significantly among land-use types and across spatial scales. The variation in the ES supply derived from land-use change has been assessed over time regionally (Zhao et al. 2004, Helian et al. 2011, Carreño et al. 2012), but few works have compared the supply of multiple ES across multiple land-use types (but see Metzger et al. 2006), and even fewer have done so at a local scale. Our work also supports previous work illustrating that the scale of analysis can alter our understanding of ES provision (Chan et al. 2006, Hein et al 2006), because the cover of land-use types can change the types and quantities of ES provided at different scales. Ecosystem services (ES) that are prominent in a small-scale analysis may be insignificant at a larger spatial scale if the land-use type responsible for their provision is scarce. For example, in our study, habitat provision in riparian forests was very important at the patch scale, but its relevance was almost negligible at the landscape scale, because of the small area riparian forests occupy in the floodplain. Likewise, climate regulation by a particular land use may seem unimportant at the patch scale, but become highly relevant when scaled up to the landscape scale because of a large area covered by that land use. Thus, the extent of any single land use at each spatial scale conditions the amount of service provided. Similarly for ES interactions, the spatial scale conditioned the scope of interactions. In our study area, only four interactions between ES remained consistent across spatial scales, highlighting the stability of some interactions. However, the majority of ES interactions changed across scales, indicating that there is no single relevant scale for analyzing ES interactions.

Although Raudsepp-Hearne et al. (2010) suggested that municipalities are a good scale at which to analyze ES interactions, in our multiscale study most ES interactions changed across spatial scales either in significance or in direction (positive vs. negative). This diversity of findings suggests that scientists and decision-makers should be aware of the spatial scales at which ES are measured and managed (Daily 2000; Carpenter et al. 2005, de Groot et al. 2010). Although as many ES and interactions as possible should be analyzed for ES and trade-offs assessments, considering at least two spatial scales is key for decision-making to assure that repercussions of management actions will stay consistent and will not reverse their effects once upscaled or downscaled. Better still, management actions should be adapted to each specific spatial scale (Aviron et al. 2009). For instance, we encountered difficulties in measuring cultural ES at the patch scale, because the information about these services is typically available at the municipality scale. Thus, data had to be downscaled, causing a potential loss of ecologically meaningful data. Moreover, given that many cultural services are influenced by municipal regulations, e.g., access to paths, recreational and fishing areas, establishment of educative panels, etc., it is advisable to measure and manage them at the municipality scale. Trying to manage such services at a large scale, e.g., landscape, may lead to disagreement among government bodies. However, ES such as provisioning services are more amenable to management at the landscape scale despite information being typically available at both patch (per hectare values) and municipal or regional scales, because they greatly influence landscape features in agroecosystems and thus, the provision of services at the landscape scale. Therefore, understanding which services respond better to each particular spatial scale is useful for ES management. Matching the appropriate scale to both ES and trade-off analyses is important when payment schemes to protect ES or to encourage sustainable agriculture are to be implemented. Studies carried out in this respect were not able to assure that schemes to enhance ES in agricultural landscapes had the same positive effects locally as regionally or at the national scale (Kleijn et al. 2006). Similarly, field-scale actions did not have the same effects locally as at the landscape scale (Concepción et al. 2012). As we have shown, they argued that this was related to the extent of land-use types under these schemes. This is especially critical when consequences of land-use policies affect millions of people (Carreño et al. 2012), such as the Common Agricultural Policy in the European Union, which incentivized agricultural intensification but has also led to a decrease in biodiversity in agricultural landscapes (Tilman et al. 2002).

We quantified the existence of trade-offs in the supply of ES, as has been posited by many authors (Rodríguez et al. 2006, Nelson et al. 2009, Laterra et al. 2012). The Millennium Ecosystem Assessment (2005) classified ES trade-offs according to their temporal and spatial scales and also depending on their reversibility and the service targeted. Although it is widely recognized that trade-offs arise because of management decisions, which derive from societal needs, values and preferences, there is little research involving societal values as a potential source of trade-offs (but see Martín-López et al. 2012). We have incorporated societal values as a likely cause of trade-offs between ES. Therefore, we classified ES interactions according to whether they can be driven by biophysical, i.e., ecological, factors or by societal values. In the first case, trade-offs are caused by biophysical interactions between ES and thus are consistent across all land uses (Table 7, example 1) or depend on the land-use type (Table 7, examples 2-4; see also Fig. 2, Table 4). Other trade-offs are caused by management decisions and are thus ultimately derived from societal values (Table 7, example 5).

We expect this classification would be applicable to other ecosystems for trade-offs analysis. Knowledge about the driving forces that provoke trade-offs can improve management for multiple ES. Biophysical trade-offs can often be reduced through specific biophysical management plans within a land-use type. For example, adequate pruning makes raw material production compatible with food production in fruit groves (Table 7, example 1) or with habitat quality in riparian forests (Table 7, example 3). Note that our results exposed the raw material production of fruit groves as a potential value, i.e., neglecting their use as fruit production. Moreover, when simultaneous gain is difficult to achieve, biophysical trade-offs can still be managed for suboptimal but compatible valuable gains (Chan et al. 2006, Trabucchi et al. 2013). Social trade-offs might be managed by considering the mix of land-use types. For example, as shown in Tables 5 and 6, most cultural services can be supplied concurrently with other ES (see also Martín-López et al. 2012).

Because of the high degree of synergies that involve cultural services, the possibility for enhancing the supply of a bundle of ES through promotion of cultural services exists in many municipalities. In our study area and probably in other river floodplains used for agricultural purposes, reopening public paths between the river and the field crops would enhance the supply of a bundle of cultural services yet causing minimal reductions in crop yield. Although synergies are more difficult to identify because significant positive correlations do not always mean that provision of one ES empowers supply of another (Table 7, examples 6-7), exploring in detail which ES or land uses correlate positively or present synergies improves the likelihood that we can enhance the total supply of ES in a targeted area. For example, promoting educational services together with recreational sites will increase the likely use of both services, enhancing the delivery of benefits to society.

Agroecosystems cover a large portion of the terrestrial surface of the Earth. As such, we cannot afford to manage them only for provisioning services because their management will condition the ES provision of the whole system. Rather they should be managed to deliver multiple ES (Bennett and Balvanera 2006, Harrison et al. 2010), enhancing especially the provision of services of those land uses covering the larger extents of the agroecosystem. To achieve this goal, research on ES compatible with agroecosystems is crucial to improve our understanding of land-use interactions (Trabucchi et al. 2012). A more comprehensive study would likely be required to set the management policies in the area. However, we can already suggest that for the Piedra River and similar floodplain agroecosystems a mosaic of habitats comprising productive crops, poplar groves, fruit groves, and restored riparian habitats would increase the supply of ES and the resilience of the floodplain ecosystem, minimizing trade-offs and creating synergies for cultural services, which could ultimately foster rural agritourism, preserve local crops and livestock varieties, promote local products, create jobs, and eventually prevent village depopulation.

Table 7. Summary of ecosystem services (ES) interactions at patch scale according to our findings.

Case	Interaction	Origin	Consistent across land uses	Example
ES 1 cannot be supplied at the same time as ES 2 in any of the land uses	Trade-off	Biophysical, ecological interactions	Yes	(1) Raw materials and food production cannot be supplied at the same time.
ES 1 is always supplied by land use *a* but ES 2 is never supplied by land use *a*	Trade-off	Biophysical, ecological interactions	No	(2) Urban areas always supply recreation but never soil stability.
ES 1 and ES 2 can be supplied at the same time at land use *a*, but never together at land use *b*	Trade-off	Biophysical, ecological interactions	No	(3) Raw materials and habitat quality can be supplied at the same time in riparian forests but never together at poplar groves.
At land use *a*, ES1 and ES2 have high values but at land use *b* ES1 have same high values and ES 2 have lower values	Trade-off	Biophysical, ecological interactions	No	(4) Raw materials and habitat quality have high values at riparian forests; at fruit groves raw materials still have high values but habitat quality is very low.
ES 1 and ES 2 can be supplied at the same time at land use *a* and land use *b*, but only sometimes appear together	Trade-off	Management option, societal decisions	Yes	(5) Food provision and fishing areas can be supplied at the same time at irrigated cereal crops and riparian forest, but only sometimes appear together.
ES 1 and ES 2 have lower values at land use *a* when they are alone than when they appear together	Synergy	Biophysical, ecological interactions	Yes	(6) Education and recreation have lower users when they are supplied alone in a location than when they are together.
ES 1 and ES 2 have medium values at land uses *a* and *b* but have always higher values at land use *c*	Synergy	Biophysical, ecological interactions	No	(7) Habitat quality and nutrient regulation have medium values at dry cereal crops and urban areas but have always higher values at riparian forests

Conclusion

Each land-use type in the Piedra River floodplain provides ES in unique quantities. Thus, preserving a mixture of land-use types is critical to providing a mixture of services. The amount of each ES supplied in a given area depends on both the per hectare provision of service in a given type of land use and the total area of each land use. The relative importance of each land-use type in supplying ES and the significant interactions among ES change depending on the spatial scale at which measurements and analysis are done. Thus, it is critical to pay careful attention to the scale of analysis considered and its impact on the conclusions. Finally, societal values, as drivers of management decisions, should be studied along with biophysical factors because they likely cause trade-offs between ES and should be considered in management plans. Uncovering the driving forces that provoke trade-offs and exploring which ES or land uses present synergies, such as those shown between cultural services in many municipalities, will enhance land managers' ability to manage ES bundles.

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Appendix 1

Detailed methods for ES sampling, indicators selected and scale adaptation.

Climate regulation

Temperature samples were recorded every 60 minutes over a period of 8 months (February to September 2012) using data loggers (iButton). Three devices per plot were hung from trees located at regular distances along a river transect perpendicular to the river channel. Three replicate plots were sampled in representative sites of each selected land use type. Dry cereal crops and Fruit groves were not surveyed but surrogate values from Abandoned crops and Poplar groves were used respectively, due to their similar cover and structure. Means of daily temperatures range (DTR=maximum temperature of day x – minimum temperature of day x) (Scheitlin and Dixon 2010) were used as an indicator. Average values by land use were calculated and used as ES values. Inverse values (1/DTR) were used to ease graphics comparisons; by doing so, higher indicator values mean higher supply of the ES (Hubbart et al. 2005; Hubbart 2011).

Gas regulation

Annual CO_2 sequestration rates were obtained from a national database (Montero 2005, CITA 2008) which estimated the amounts of carbon stored by the above- and below-ground biomass of different plant species. This database calculates the species annual growth and transforms it into carbon equivalent tons per hectare, considering that Carbon stored = Biomass x 0.4735. Then carbon is transformed into CO_2 using their stoichiometric coefficients. Grass species are not supposed to store any CO_2, because they grow up, die and get decomposed annually, and thus their annual balance equals zero (CITA 2008).

Soil stability

The organic matter on topsoil (0-10 cm) was differentiated visually and its depth recorded with a measuring tape. Leaf litter was excluded. Three samples were taken along a river transect perpendicular to the river channel and three transects replicates were taken in each plot. Three plot replicates were sampled by land use except for urban soils. Soils were first surveyed in September 2010 and replicated in July 2011 and 2012.

Nutrient regulation

Soil samples (0.5 kg) were taken from the topsoil (0 – 15 cm). Three samples were taken along a river transect perpendicular to the river channel and three transects replicates were taken in each plot. Three plots replicates were sampled by land use except for urban soils. Soils were dried, sieved and milled prior to lab analyses. Total Nitrogen was measured using a macro elemental analyzer (Vario Macro Max CN) and results were expressed in concentration (ppm). Soils were first surveyed in September 2010 and replicated in July 2011 and July 2012.

Habitat quality

Habitat quality was evaluated using the Riparian Quality Index (González del Tánago et al. 2006). Three replicate plots per land use were sampled during the field campaigns between July 2010, July 2011 and July 2012.

Raw materials

Annual growth rates per plant species were obtained from a national database (Montero 2005, CITA 2008) which calculated the annual growth as tons of biomass tons per hectare, considering that Biomass = Correction factor x timber diameter. Grass species are not supposed to accumulate any biomass annually, because they grow up, die and get decomposed, and thus the annual balance equals zero (CITA 2008). Other woody species and woody formations were calculated individually by plant species.

Food production

Yield values for the crops growing within the study area were obtained from national databases statistics (Instituto Nacional de Estadística, updated on 30 October 2012) expressed as kilograms per hectare and multiplied by the crop caloric value (kilocalories per 100 grams). The ES value is expressed as kilocalories per hectare.

Fishing

Available fishing stretches for recreational use at the river Piedra were obtained from the fishing regulatory policy of 2012 for the Autonomous Community of Aragon (BOA 2012) and drawn using GIS tools (ArcGIS 10.0, ESRI). Fishing available stretches were computed for both riversides. Stretches were converted into polylines, their perimeters calculated and summarized into stretches available or unavailable for fishing. Polylines were converted into polygons and intersected to the land use cover with a buffer of 10 m to add both the land use and municipality information of each stretch of the river. Then lengths were recalculated. Total length across river stretches of each land use type was used as an indicator at landscape scale. Average values by land use type across municipalities were used as an indicator at landscape scale. The length of the river across each land use type in relation to the total length of the river (i.e. including areas unavailable for recreational fishing) and in reference to a 1 hectare patch (a patch of 100 meters of side) was used as an indicator at patch scale (i.e. Fishing at land use x = (Total length of land use x / Total length of the river)*100).

Sports

Tracks of post-signed and user-designed paths were downloaded from both the local tourist office website and wikilocs (http://senderos.turismodearagon.com and www.wikiloc.com, respectively; date of reference: 12 October 2012) following Trabucchi et al. (2013b). Tracks around the study area were unified using GIS tools (QGIS, Quantum GIS Development Team), and overlapped to the study area viewshed. Then the viewshed of the shapefile obtained was calculated and intersected to the land use cover. Finally the extent of each land use that can be seen from the open-to-public used paths was calculated. Average values per hectare of each land use were used at patch scale. Values at municipality and landscape scale were obtained directly from the GIS attribute table in hectares.

Recreation

The number of areas for social amenity per land use and municipality were counted in situ in all the municipalities in August 2012. The average number of rest areas per hectare of land use was estimated as the total number of rest areas of each land use in the study area divided by the total number of hectares that each particular land use covers within the study area.

Education

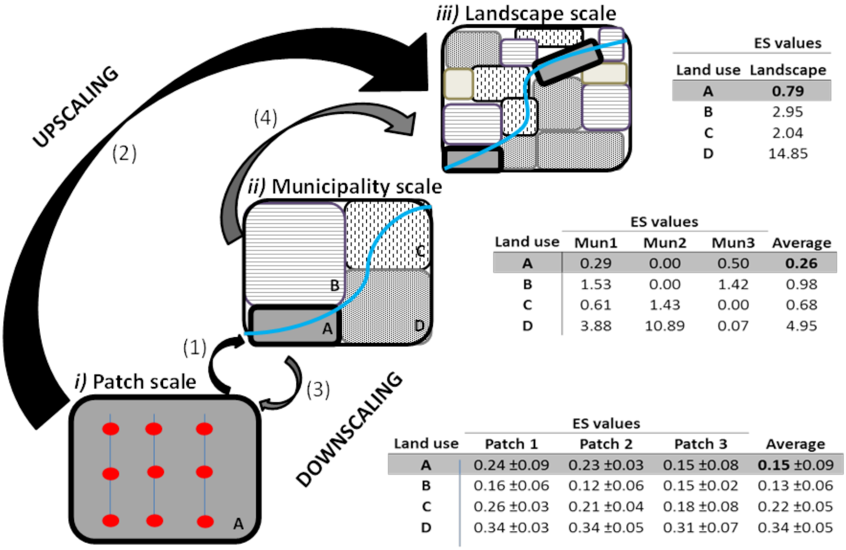
The number of notice boards with information about the ecosystem of the study area was counted in situ per land use and municipality in August 2012. To calculate the average number of notice boards per hectare of land use, the total number of notice boards of each land use was divided by the total number of hectares that each particular land use occupies within the study area.

Social relationships

The number of local associations related to the use of nature (either for conservation or for agriculture) per municipality within the study area was counted in August 2012. Downscaling to patch scale was estimated as follows,

$$Assoc_x = \frac{\sum_{i=1}^{n}(\frac{A_i}{S_i} * X_i)}{n}$$

where AssocX is the average number of local associations related to the use of nature at land use x; A is the number of local associations at municipality i; S are hectares of municipality i within the study area; X are hectares of land use x within the study area at municipality i; and n is the number of municipalities.



Land use	ES values Landscape
A	**0.79**
B	2.95
C	2.04
D	14.85

Land use	ES values Mun1	Mun2	Mun3	Average
A	0.29	0.00	0.50	**0.26**
B	1.53	0.00	1.42	0.98
C	0.61	1.43	0.00	0.68
D	3.88	10.89	0.07	4.95

Land use	ES values Patch 1	Patch 2	Patch 3	Average
A	0.24 ±0.09	0.23 ±0.03	0.15 ±0.08	**0.15** ±0.09
B	0.16 ±0.06	0.12 ±0.06	0.15 ±0.02	0.13 ±0.06
C	0.26 ±0.03	0.21 ±0.04	0.18 ±0.08	0.22 ±0.05
D	0.34 ±0.03	0.34 ±0.05	0.31 ±0.07	0.34 ±0.05

Land use	Cover (ha) Mun1	Mun2	Mun3	Total
A	1.91	0.00	3.31	**5.22**
B	11.78	0.00	10.89	22.67
C	2.77	6.50	0.00	9.27
D	11.41	32.04	0.22	43.67

(1) $$\frac{\sum_{i=1}^{n} \begin{matrix}(Average\ ES\ value\ at\ the\ patch \\ x \\ Cover\ of\ land\ use\ A\ at\ municipalityi)\end{matrix}}{n\ municipalities}$$

(2) $$\begin{matrix}Average\ ES\ value\ at\ the\ patch\ x \\ \sum_{i=1}^{n} Cover\ of\ land\ use\ A\ at\ municipalityi\end{matrix}$$

(3) $$\frac{\sum_{i=1}^{n} (ES\ value\ at\ municipalityi / Cover\ of\ land\ use\ A\ at\ municipalityi)}{n\ municipalities}$$

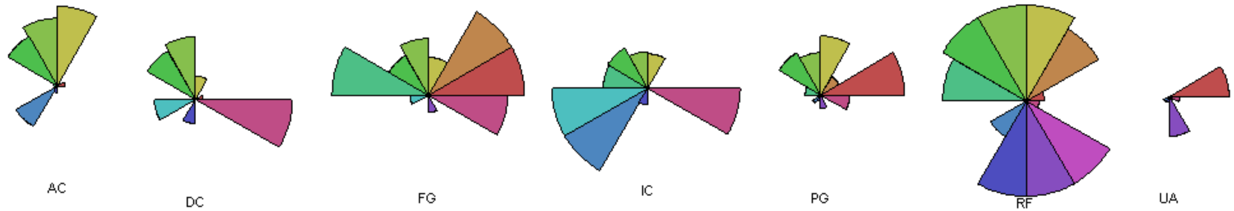
(4) $$\sum_{i=1}^{n} ES\ value\ at\ municipalityi$$

Figure S1. ES scaling methods. The figure represents the scaling techniques used to obtain the values of ES provision by land use types (land use A, as an example) at three spatial scales (patch, municipality, and landscape). Assessment of regulating, supporting and provisioning services (black arrows, case 1) starts at the patch scale, and assessment of cultural services (greys arrows, case 2) starts at the municipality scale. In case 1 (e.g. Nutrient regulation), we sampled three patches per land use type (figure *i)* shows patch 1 of land use A) collecting 9 soil samples (red dots) per patch to analyze total nitrogen content as a proxy of the ES. We averaged values within each patch and across the three patches of land use A to obtain a unique value of the ES at the patch scale. To scale up to municipality (small black arrow, equation (1)), we used the average value of land use A at the patch scale (i.e. 0.15) and multiplied it by the cover of land use A at municipality 1 (Mun1, see Land cover table on the right; i.e. 0.15x1.91=0.29). We did the same with each municipality and averaged values to obtain a single value of ES provision by land use A at the municipality scale (i.e. 0.26). We started from the patch scale again to scale up to landscape (large black arrow, equation (2)). We used the average value of the ES for land use A and multiplied it by the total cover of land use A in the landscape (i.e. the sum of the cover of land use A across the three municipalities; 0.15x5.22=0.78). In case 2 (e.g. Education), we counted the number of educational sites within land use A in each municipality (figure *ii)* shows municipality 1). We scaled down to the patch (small grey arrow, equation (3)) dividing the ES value of each municipality by the cover of land use A in that municipality (e.g. for municipality 1, 0.29/ 1.91=0.15). We averaged these values across the three municipalities to obtain a single value of ES provision by land use A at the patch scale (i.e. (0.15+0.00+0.15)/3=0.1). To scale up to the landscape (large grey arrow, equation (4)), we summed the ES values at land use A across the three municipalities (i.e. 0.29+0.00+0.50=0.79).

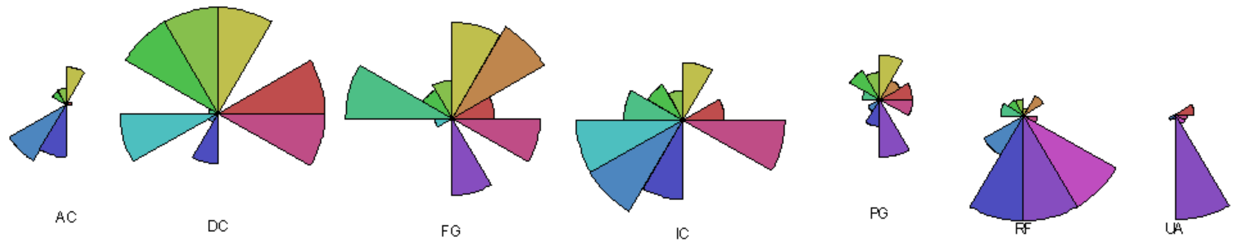
Appendix 2

Detailed information about the supply of ES by each land use across spatial scales.

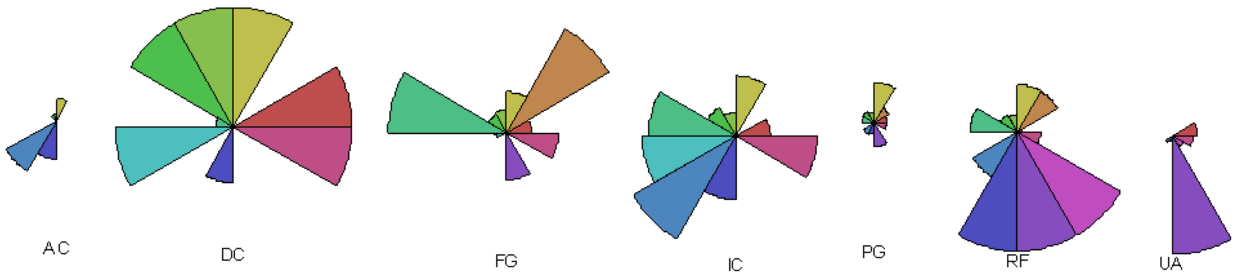
Patch scale



Municipality scale



Landscape scale



Reference

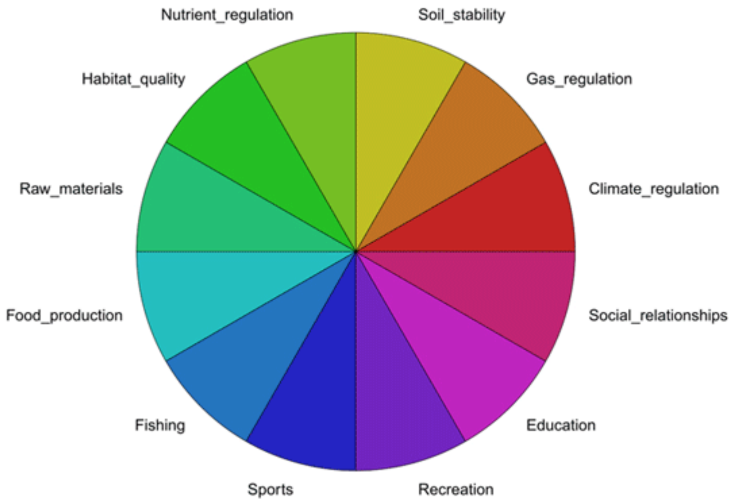


Fig. S2. Relative contribution to the provision of ES by land use type at each spatial scale. Pie size represents the relative value in relation to the reference pie chart (the maximum value of the data). Note that empty slices represent the minimum relative value to the contribution of that particular ES. (Abbreviations: AC=abandoned crops; DC=dry cereal crops; FG=fruit groves; IC=irrigated cereal crops; PG=poplar groves; RF=riparian forest; UA=urban areas).

Table S1. Ecosystem services delivered based on the estimated proportion of each land use type at each spatial scale (patch, municipality and landscape). (Abbreviations: AC=abandoned crops; DC=dry cereal crops; FG=fruit groves; IC=irrigated cereal crops; PG=poplar groves; RF=riparian forest; UA=urban areas).

Scale / Land use	Climate regulation	Gas regulation	Soil stability	Nutrient regulation	Habitat quality	Raw materials	Food production	Fishing	Sports	Recreation	Education	Social relationships
	$ºC^{-1}$	$C0_2$eqTons/Ha	Cm	Ppm	Score	Tons/Ha	Kcal/Ha	m	Ha	Items	Items	Number
Patch												
AC	0.015	0.637	0.303	0.066	13.423	1.169	0.000	5.486	0.415	0.000	0.000	0.034
DC	0.009	0.000	0.049	0.037	7.357	0.453	1589132.782	0.124	0.775	0.000	0.000	0.183
FG	0.004	9.198	0.029	0.014	2.190	5.298	259907.184	0.001	0.010	0.001	0.000	0.056
IC	0.004	0.000	0.033	0.011	2.750	3.040	1695297.085	2.803	0.251	0.000	0.000	0.080
PG	0.006	3.264	0.073	0.015	3.687	1.393	0.000	0.373	0.128	0.001	0.000	0.039
RF	0.011	27.372	0.258	0.071	15.901	15.842	0.000	3.356	3.685	0.936	0.144	0.052
UA	0.004	0.000	0.000	0.000	0.456	0.000	0.000	0.178	0.081	0.003	0.002	0.015
Municipality												
AC	0.000	0.021	0.010	0.002	0.434	0.038	0.000	0.177	0.013	0.000	0.000	0.001
DC	0.019	0.000	0.112	0.083	16.636	1.023	3593165.719	0.281	1.753	0.000	0.000	0.413
FG	0.005	12.210	0.039	0.019	2.907	7.032	345005.039	0.001	0.014	0.001	0.000	0.074
IC	0.010	0.000	0.089	0.029	7.329	8.102	4518511.742	7.472	0.669	0.000	0.000	0.212
PG	0.002	1.297	0.029	0.006	1.465	0.553	0.000	0.148	0.051	0.001	0.000	0.016
RF	0.003	6.360	0.060	0.016	3.694	3.681	0.000	0.780	0.856	0.217	0.033	0.012
UA	0.003	0.000	0.000	0.000	0.401	0.000	0.000	0.157	0.071	0.002	0.002	0.013
Landscape												
AC	0.001	0.045	0.021	0.005	0.950	0.083	0.000	0.388	0.029	0.000	0.000	0.002
DC	0.020	0.000	0.118	0.088	17.660	1.087	3814539.896	0.299	1.861	0.000	0.000	0.439
FG	0.005	12.700	0.041	0.020	3.023	7.315	358857.140	0.001	0.014	0.001	0.000	0.077
IC	0.007	0.000	0.060	0.020	4.896	5.413	3018947.437	4.992	0.447	0.000	0.000	0.142
PG	0.003	1.807	0.040	0.008	2.042	0.771	0.000	0.206	0.071	0.001	0.000	0.022
RF	0.002	5.008	0.047	0.013	2.909	2.898	0.000	0.614	0.674	0.171	0.026	0.009
UA	0.005	0.000	0.000	0.000	0.617	0.000	0.000	0.242	0.109	0.004	0.003	0.020

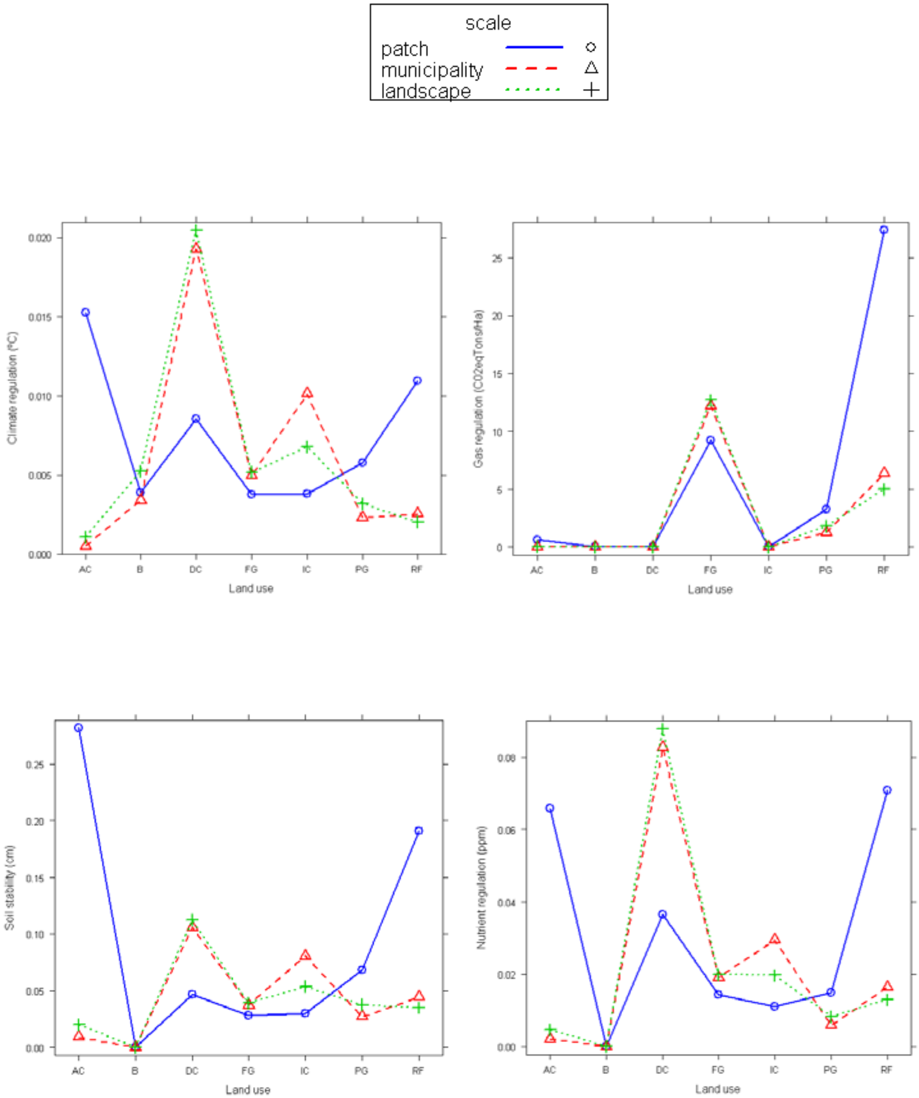
Table S2. Comparison across spatial scales of the supply of ES per hectare by each land use type. See Fig. S2 for details.

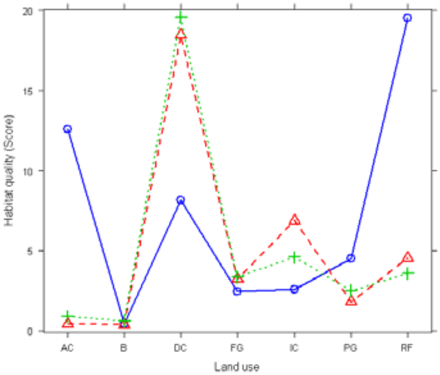
Scale	Sumary
Patch	Riparian forest was the land use supplying most ES per hectare and with the highest values per hectare: gas regulation, nutrient regulation, habitat quality, raw materials, sports, recreation and education. Abandoned crops supplied the most climate regulation, soil stability and fishing areas, irrigated cereal crops produced the most part of food whereas dry cereal crops comprised most social relationships.
Municipality	Dry cereal crops were the main supplier of climate regulation, soil stability, nutrient regulation, habitat quality, sports and social relationships per hectare; fruit groves supplied the most of gas regulation; irrigated cereal crops supplied the most of food, raw materials and fishing; and riparian forest was the main supplier of recreation and education per hectare.
Landscape	Most ES per hectare were supplied by dry cereal crops (climate regulation, soil stability, nutrient regulation, habitat quality, food production, fishing areas, sports and social relationships). Minor contributors were riparian forests (recreation and education), fruit groves (gas regulation and raw materials).

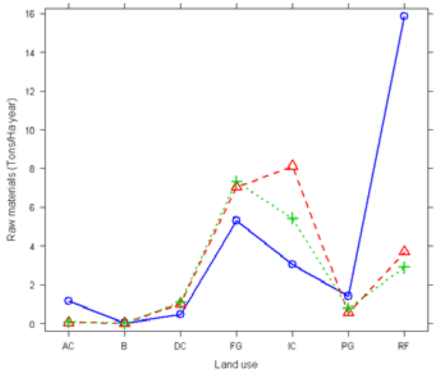
Table S3. Significant factors in the supply of ES per hectare across three spatial scales. Significance codes: '***' for 0; '**' for 0.001; '*' for 0.01; '.' for 0.05.

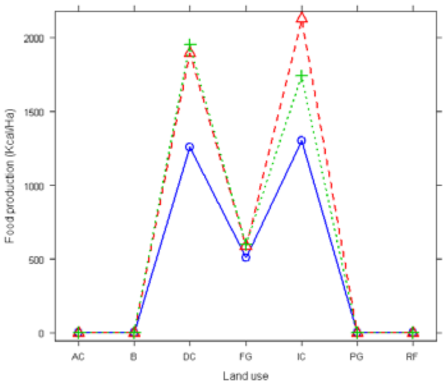
Ecosystem service	Term	Chisq	Df	Pr(>Chisq)	Significance
Climate regulation	Land use	0.34	6	1.00	
	Scale	0.00	2	1.00	
	Land use-Scale	0.27	12	1.00	
Gas regulation	Land use	85.83	6	2.23E-16	***
	Scale	25.77	2	2.53E-06	***
	Land use-Scale	39.91	12	7.46E-05	***
Soil stability	Land use	6.71	6	0.35	
	Scale	5.03	2	0.08	.
	Land use-Scale	14.60	12	0.26	
Nutrient regulation	Land use	4.71	6	0.58	
	Scale	0.56	2	0.76	
	Land use-Scale	3.41	12	0.99	
Habitat quality	Land use	177.97	6	<2e-16	***
	Scale	2.64	2	0.268	
	Land use-Scale	116.60	12	<2e-16	***
Raw materials	Land use	81.36	6	1.87E-15	***
	Scale	5.77	2	0.06	.
	Land use-Scale	41.11	12	4.70E-05	***
Food production	Land use	1.10	10	1.00	
	Scale	27.16	6	0.00	***
	Land use-Scale	0.16	12	1.00	
Fishing	Land use	56.92	6	1.90E-10	***
	Scale	2.86	2	0.24	
	Land use-Scale	24.21	12	0.02	*
Sports	Land use	23.31	6	0.00	***
	Scale	0.57	2	0.75	
	Land use-Scale	7.83	12	0.80	
Recreation	Land use	0.58	8	1.00	
	Scale	2.57	4	0.63	
	Land use-Scale	0.02	12	1.00	
Education	Land use	0.19	8	1.00	
	Scale	0.31	4	0.99	
	Land use-Scale	0.01	12	1.00	
Social relationships	Land use	4.03	6	0.67	
	Scale	0.22	2	0.90	
	Land use-Scale	0.52	12	1.00	

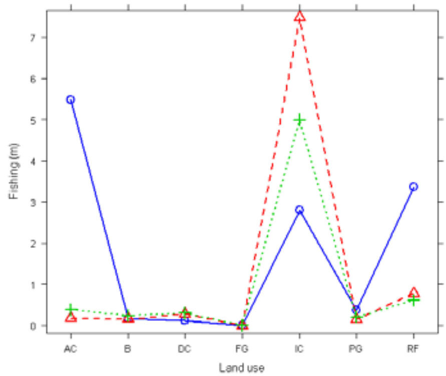
Fig. S3. Comparison in the supply of ES per hectare by each land use at three different spatial scales: patch, municipality and landscape. Horizontal axis shows land uses and vertical axis shows ES indicators (numbers express relative values per hectare). Lines are provided to improve scale differentiation. Note that most ES were delivered at patch scale, and only food production and social relationships were supplied mainly at municipality scale. Abbreviations: AC=abandoned crops; DC=dry cereal crops; FG=fruit groves; IC=irrigated cereal crops; PG=poplar groves; RF=riparian forest; UA=urban areas.

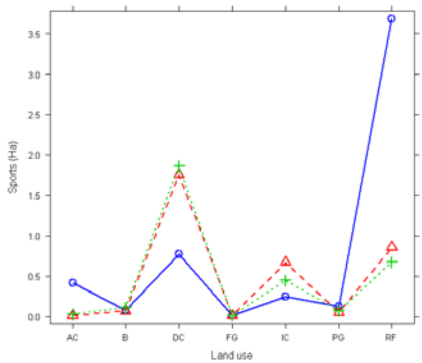


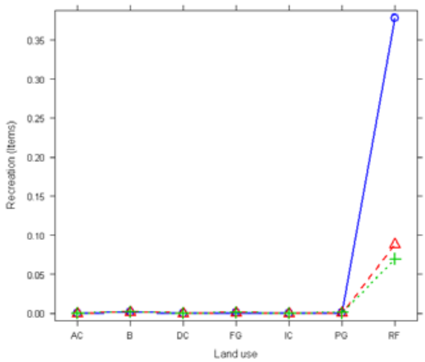

Habitat quality (Score)
0
5
10
15
20
AC
B
DC
FG
IC
PG
RF
Land use

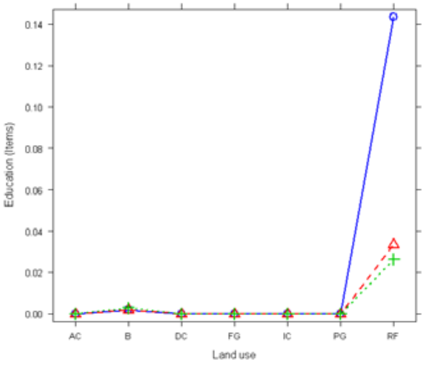

Raw materials (Tons/Ha year)
0
2
4
6
8
10
12
14
16
AC
B
DC
FG
IC
PG
RF
Land use

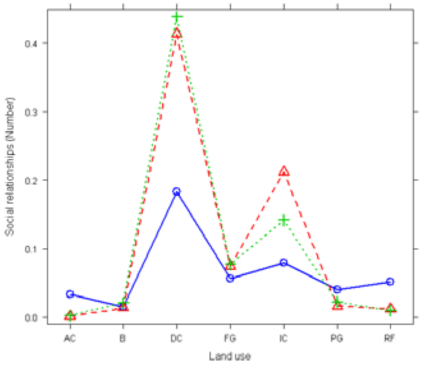

Food production (Kcal/Ha)
0
500
1000
1500
2000
AC
B
DC
FG
IC
PG
RF
Land use


Fishing (m)
0
1
2
3
4
5
6
7
AC
B
DC
FG
IC
PG
RF
Land use


Sports (Ha)
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
AC
B
DC
FG
IC
PG
RF
Land use


Recreation (items)
0.00
0.05
0.10
0.15
0.20
0.25
0.30
0.35
AC
B
DC
FG
IC
PG
RF
Land use


0.14
0.12
0.10
0.08
0.06
0.04
0.02
0.00
Education (Items)
AC
B
DC
FG
IC
PG
RF
Land use


0.4
0.3
0.2
0.1
0.0
Social relationships (Number)
AC
B
DC
FG
IC
PG
RF
Land use

CAPÍTULO 5. A FRAMEWORK FOR THE SOCIAL VALUATION OF ECOSYSTEM SERVICES*

ABSTRACT. Methods to assess ecosystem services using ecological or economic approaches are considerably better defined than methods for the social approach. To identify why the social approach remains unclear, we reviewed current trends in the literature. We found two main reasons: (i) the cultural ecosystem services are usually used to represent the whole social approach, and (ii) the economic valuation based on social preferences is typically included in the social approach. Next, we proposed a framework for the social valuation of ecosystem services that provides alternatives to economics methods, enables comparison across studies, and supports decision-making in land planning and management. The framework includes the agreements emerged from the review, such as considering spatial–temporal flows, including stakeholders from all social ranges, and using two complementary methods to value ecosystem services. Finally, we provided practical recommendations learned from the application of the proposed framework in a case study.

Key words: Social evaluation; Stakeholder; Ecosystem services flow; Ecosystem services ranking; Social perception.

* Felipe-Lucia MR, FA Comín, J Escalera-Reyes. 2014. A framework for the social valuation of ecosystem services. *AMBIO* 1–11. DOI: 10.1007/s13280-014-0555-2

Introduction

The use of ecosystem services [the benefits humans receive from nature (Alcamo et al. 2003)] is becoming a powerful tool in land planning and management. According to the subject of study to be valuated, the study of ecosystem services can be approached from an ecological, economic, or social perspective. The ecological approach focuses on measuring ecological functions or ecosystem properties (de Groot et al. 2002); the economic approach estimates the use and non-use values of ecosystems in monetary terms (Wilson and Carpenter 1999); and the social approach is based on the values society attributes to each ecosystem service (Martín-López et al. 2012). However, the unclear existing methodology to assess ecosystem services from the social approach (Menzel and Teng 2010) is risking the potential impact of the ecosystem services framework in land planning and management (Chan et al. 2012a). For instance, the fringe between the economic and the social approach is not well distinguished, leading to the frequent use of econometric methods to assess social preferences on ecosystem services. In other instances, the social approach is only implemented to assess cultural ecosystem services, disregarding the rest of the services (such as regulating, supporting, and provisioning) (Newton et al. 2012; Plieninger et al. 2013). The omission of the other types of services in the social valuation of ecosystem services might be due, among other reasons, to the expertise and amount of time that these methods require, and to the usual confusion between the category of socio-cultural ecosystem services [i.e., "the nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences"; Millennium Ecosystem Assessment (MEA) 2005, p. 40] and the social approach of ecosystem services (which evaluates all ecosystem services).

In ecosystems management, social valuation has typically been implemented with the aim of achieving policy makers' objectives [e.g., river restoration projects and water and natural-resource management (Menzel and Teng 2010)]. However, its potential can be extended further by including the participation of society in ecosystem services assessments advising decision-making (Chan et al. 2012a). This will more likely enable legitimate results and satisfactory decisions to more stakeholders (Menzel and Teng 2010). In turn, that will help to develop more resilient communities (Folke et al. 2002) built on social fulfillment and environmental sustainability (Castillo et al. 2005; Berkes and Turner 2006).

Developing a framework to guide social assessments of ecosystem services is a challenge where collaboration between social and natural scientists is required (Maass et al. 2005; Raymond et al. 2013). Yet to our knowledge, this challenge has not been addressed, and several approaches can be pursued. Here, we apply multiple disciplines that influence the expression of ecosystem services preferences by stakeholders (e.g., anthropology, sociology, and psychology), together with views of experts on ecosystem management to devise such a framework. We aim to use this as a common ground to share expertise across social assessments of ecosystem services, and to support land planning and management. As we will show in this article, such comparisons across studies are currently limited by incomparable spatial and temporal scales, disparate methods of evaluating ecosystem services, and especially by the different status of stakeholders involved.

The objectives of this paper are: (1) to explore how the social valuation of ecosystem services has been addressed to date in the scientific literature, (2) to propose a novel framework to guide social valuations of ecosystem services, and (3) to illustrate the proposed framework via a case study.

Methods

To develop a framework to guide social valuations of ecosystem services, we first explored how the social valuation of ecosystem services has been addressed to date through an in-depth literature review; secondly, we proposed a framework including aspects that emerged from the review; and thirdly, we implemented the proposed framework in a case study. Below, we describe the methods used in each part.

Current trends in the social valuation of ecosystem services

To comment on the current trends relative to the social valuation of ecosystem services and to identify why this approach remains unclear, we reviewed all articles found across all type of sources (i.e., journals, conference proceedings, and books or book chapters) indexed in the ISI Web of Knowledge (which included the Web of Science, Medline, Zoological Records, and the Journal of Citation databases) published before the end of September 2013, that contained the keywords “ecosystem services”, and either the keywords “social valuation”, “preferences” or “stakeholders” in the title or topic. We obtained a total of 1082 records (214, 328, and 540 records in each search, respectively). We checked their suitability by reading the title and abstract, or reading the article in full. After rejecting double-counting papers, records not published in English, papers that did not explicitly undertake a social evaluation of ecosystem services (for example, papers proposing methods, frameworks, or reviews), and papers assessing social preferences on ecosystem services solely by economic methods, 55 records remained (see the list of selected papers in Electronic Supplementary Material, S1). The remaining articles were carefully read, and the targeted information was extracted to calculate percentages of each aspect addressed.

A framework for the social assessment of ecosystem services

To develop a framework to guide social assessments of ecosystem services, we focused on the basic questions required: *Who should complete the evaluation?, How to focus it?, At what extent?,* etc. We incorporated each question as a stage in the assessment that can be more thoroughly examined if taken as an iterative process (Fig. 1).

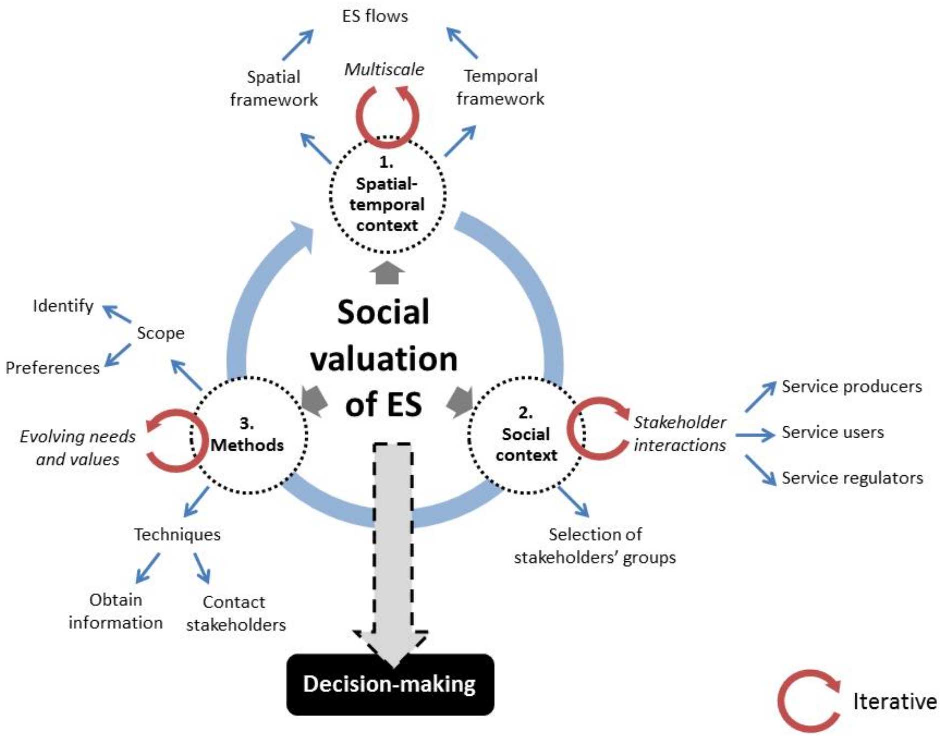


Figure 1. Framework for the social valuation of ecosystem services (ES). According to time and funding availability, all stages can be used in an iterative process to help decision-making. In the first stage, the spatial–temporal context is first broadly defined, and is then expanded to a multiscale assessment in second or successive rounds. In the second stage, the stakeholders selected to represent the social context can be more exhaustively detailed to identify the interactions among them. In the third stage, the appropriate method can be iteratively applied to reflect evolving preferences and views.

<u>Stage 1. The spatial and temporal context</u>

Once we have elucidated the aim of the project–what is to be assessed–delimiting the spatial and temporal boundaries is the first step toward evaluating ecosystem services (Hein et al. 2006; Chan et al. 2012a). Ideally, the study area should be extended to include the causes and effects on the object of study, but in practice, it is sufficient to limit it to the timespan and territories that influence both the biophysical and the sociological dimensions the most. Since the appreciation of ecosystem services hinges on stakeholders' dependence and their preferences might change over time and across spatial scales (Alcamo et al. 2003; Turner et al. 2003; Hein et al. 2006; Lamarque et al. 2011), a multiscale assessment of ecosystem services is valuable (Trabucchi et al. 2013). This process might increase the complexity of the evaluation, but capturing a greater variety of opinions and interactions among stakeholders and the ecosystem also increases knowledge concerning the decision context and enables the adaptation of management policies to each spatial and temporal scale (Hauck et al. 2013).

Stage 2. The social context

Who should evaluate ecosystem services? Ideally, all stakeholders of the project [i.e. the population that has a real influence on the object of study, or that might be affected by decisions made concerning it (Freeman 2010)] should participate (Satz et al. 2013). Stakeholders' opinions can be requested from a single person, a sample of citizens, and the involvement of the total population (Antunes et al. 2009). More practically, stakeholders are usually grouped to ultimately include a small fraction of them (i.e., the key players; Chan et al. 2012a). Stakeholders that are required to express their opinions can be clustered by a myriad of criteria (age, sex, place of residence, profession, education, economic level, and political or religious beliefs), of which each might assign different values to ecosystem services (Cowling et al. 2008) depending on their views and needs (Vermeulen and Koziell 2002). As the social valuation of ecosystem services is intended to guide decision-making on ecosystem services management, it might be more convenient to group stakeholders according to their use of the ecosystem (e.g., irrigators, walkers, and conservationists) and their role in the government and social life of the area. With a good representation of stakeholders, outcomes are more likely to represent the actual values of the targeted area, avoiding trends of what are important ecosystem services to evaluate (Castillo et al. 2005; Escalera Reyes 2011; Moreno et al. 2014).

Stage 3. The methods for social assessment

Methods to elicit social preferences are varied, and depend on the scope of the study. Most studies focus on identifying valuable ecosystem services of an area (Maass et al. 2005), others aim to rank the importance of such services (García-Llorente et al. 2012), and some reflect evolving human preferences and views through time (Aretano et al. 2013). Choosing a particular method might influence the results, but combining several methods according to our objectives might capture opinions from a broader spectrum, avoiding possible bias. In general, qualitative methods (see

Chan et al. 2012a) are more useful for assessing ecosystem services because they enable a comprehensive understanding of the interactions between humans and the ecosystem (Daniel et al. 2012). Moreover, the most effective way to contact stakeholders and the methods used to analyze their responses are also important matters, the choice of which depends on the type of stakeholder approached.

In addition, considering ecosystems from the perspective of each stakeholder or beneficiary (Ringold et al. 2013) makes it easier to differentiate between the valuation of the service (what is supplied to the beneficiary) and the value given to it [what is weighted by the beneficiary (Tallis et al. 2012)]. Furthermore, a previous understanding of the reasons why an ecosystem service is valued is essential for comparing valuation outcomes across studies (see examples of typologies of values in Hein et al. 2006; Anthony et al. 2009; and Chan et al. 2012b).

Implementing the framework in a case study: The River Piedra floodplain

To illustrate the implementation of the framework proposed, we undertook a social valuation of the ecosystem services of the River Piedra floodplain (Spain). In this case study, we aimed to analyze whether the different perceptions of ecosystem services among stakeholder groups were related to their use of the ecosystem–were related to their main economic and leisure activities.

Spatial and temporal context

The spatial boundary was limited to the floodplain of the River Piedra (19.3 km^2), a homogeneous area where the inhabitants depend on the riparian ecosystem for daily activities such as farming, nature tour operators, or visiting a natural waterfall park. The interviews provided information about the ecosystem services flows in the area over the last 50 years, but the ranking of ecosystem services preferences was based on the present. However, defining the temporal framework in the present was not easy to clarify; instead of ranking ecosystem services independently of what is currently delivered, some stakeholders ranked their preferences according to their perception of what is being currently delivered. To ensure consistency, these latter responses were rejected.

Social context

From a total population of 880, we contacted 71 people in person, including permanent and temporal residents, farmers, tour operators (hosting or guiding nature tourists), nature protection agents, scientists, and technicians working on riverbank restoration projects. Some of these people, such as local mayors and regional[1] authorities, were contacted because of their relevant social role in decision-making, and in influencing perceptions about the river and the floodplain (i.e., local pro-environmental associations).

Methods of assessment

We performed semi-structured interviews for a qualitative sample of the main stakeholders of the River Piedra floodplain. Interviews were mostly held individually and occasionally in groups of two or three people from the same stakeholder sector (namely, when new stakeholders were contacted on site) and lasted from 30 to 90 min. Digital records of interviews were kept with the interviewees' agreement. A minimum number of seven people from each of the main stakeholder sectors were interviewed; until we did not receive more information from the same sector of stakeholders (Valles 1999). This method maximizes the survey effort by obtaining a wide range of different answers. We were interested in both ecosystem services identification and preference rankings. Therefore, in the first part of the interview, we asked about the uses, products, and benefits that the interviewees derived from the River Piedra and how these had changed over the last 50 years. In the second part, we provided the interviewees with a list of 21 benefits derived from the River Piedra and asked them to rank the services according to what they considered more important for maintaining their standard of living (see the list of cards in Electronic Supplementary Material, S2).

Results

The first section shows the results of the review, organized according to the stages of the framework proposed. In the second section, we roughly explain the outcomes obtained from the implementation of the proposed framework in the River Piedra case study.

Current trends in the social valuation of ecosystem services

Stage 1

Spatial framework: The results of our review showed that most evaluations (40.6 %) occurred at a supra-local scale, larger than the municipality (i.e., county, province). The rest of spatial scales were addressed in a downscaling order as follows: region (a continent or a part of one) (1.6 %), state or country (3.1 %), small islands (3.1 %), watershed or valley (20.3 %), municipality (29.7 %), and farm (1.6 %) (Fig. 2a). In addition, the most-studied ecosystems (classification based on the MEA 2005) were cultivated (34.6 %), forest (24.7 %), inland water (11.1 %), dryland (8.6 %), mountain (7.4 %), coastal (6.2 %), island (3.7 %), and urban systems (3.7 %). Studies considering marine and polar ecosystems were not found in our review.

Temporal framework: Eighty percent of studies focused on current service provision, whereas only 9 % were based on a comparison between past and current provision, and 7.2 % compared present and future expectations (Fig. 2b). Finally, 1.8 % of studies projected future ecosystem service provision, and another 1.8 % compared the provision of services across past, present, and future ecosystem services scenarios.

Stage 2

Social context: From our review, 38.3 % of the studies considered the opinions of local residents, 25.2 % consulted local or regional[1] representatives (including mayors, NGOs, and major associations), and 17.8 % included environmental professionals such as scientists and technicians. National authorities were considered in 9.4 % of the studies, and 7.5 % included the views of visitors or tourists (Fig. 3). Thirty-eight percent of studies were based exclusively on a single stakeholder group; namely, 29 % of studies were addressed to local inhabitants, 5.5 % to local or regional representatives, and 3.6 % to experts. No study relied solely on the opinion of national authorities, and the rest considered a mixture of several types of stakeholders. A small number of studies compared views between two stakeholder groups, for example, locals versus visitors, landowners versus tenants, and permanent residents versus seasonal ones.

Stage 3

Scopes: Our review revealed two scopes for evaluating ecosystem services, and both were used equally: 34.5 % of the evaluations focused on identifying ecosystem services (asking participants to elaborate a list of services to test their environmental knowledge), 34.5 % focused on establishing preferences among ecosystem services (asking participants to sort ecosystem services according to their priorities), 27.3 % of the studies considered both scopes, and 3.6 % used social evaluation to elicit uniquely cultural services (Fig. 4a).

Techniques: In our review, 34.3 % of the studies used discourse analysis, 27.1 % used Likert-type scales [a measure of the level of agreement or disagreement to a statement according to a symmetric scale; e.g., 1–5, 0–3, 0–10 (Likert 1932)], 22.9 % used ranking or weighting [including AHP (Analytical Hierarchy Process) (Saaty 1980) and swing-weighting], 8.6 % used Multi Criteria Decision Aid [MCDA (Belton and Stewart 2001)], 5.7 % used community mapping, and 1.4 % used outcomes from a workshop or focus group (Fig. 4b). However, the majority of studies were based on a single methodology; primarily, discourse analysis (24 %), the Likert-type scale (24 %), and ranking or weighting (13 %). The combination of discourse analysis and ranking or weighting was used in 11 % of studies. Additionally, 16.4 % of the valuations included some type of economic valuation. Finally, our review showed that almost half of the valuations (46.8 %) included interviews (97 % were held face-to-face), 29 % organized workshops or focus groups, and 22.6 % distributed surveys (including face-to-face and by mail) (Fig. 4c). Eighty percent of studies were based exclusively on a single approach; namely, 35 % of the valuations were accomplished uniquely through face-to-face interviews, 22 % through surveys, and 22 % as workshops or focus groups. A minority of the valuations (1.6 %) were completed entirely by an expert panel and by e-mail, and the rest considered a mixture of techniques.

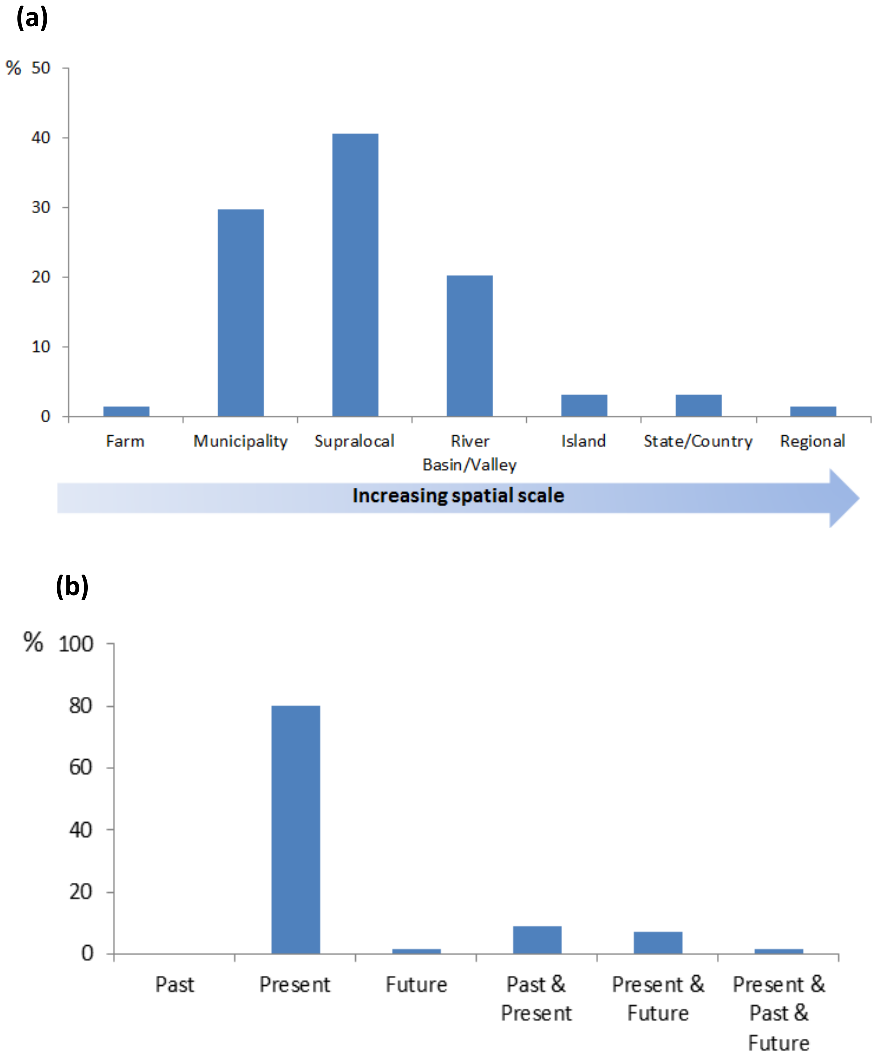


Figure 2. Percentage of valuations accomplished on different (a) spatial and (b) temporal scales. Note that in (a) *supra-local* refers to a scale larger than municipality (i.e., a county or province) and *regional* refers to a continent or a part of one.

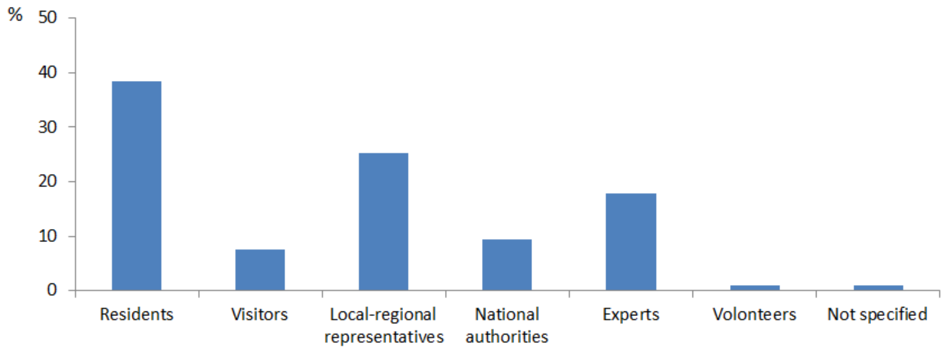


Figure 3. Social context: percentage of types of stakeholders asked to evaluate ecosystem services. Note that *local–regional* representatives include mayors, NGOs, and major associations of a county or province; and *experts* refers to environmental professionals (scientists and technicians).

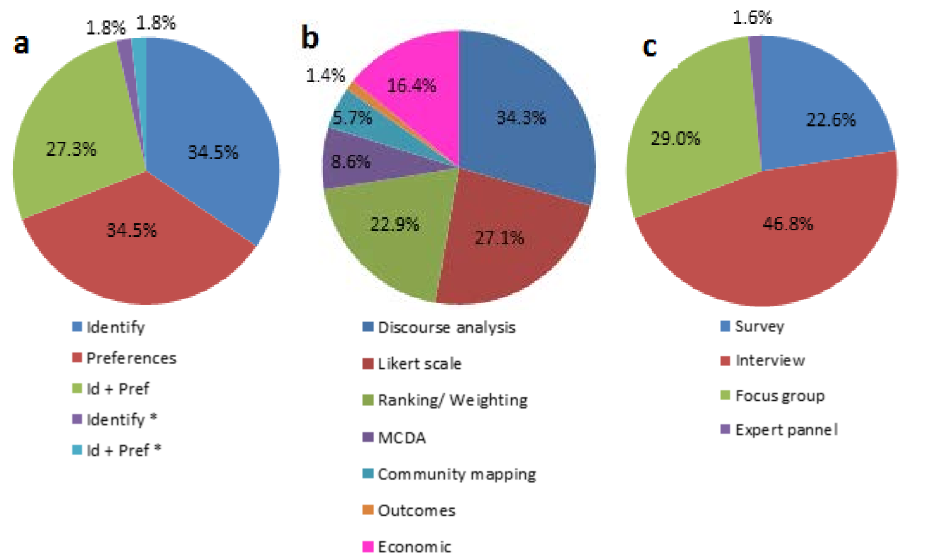


Figure 4. Percentage of each method used (a) as scope, (b) to analyze the social valuation of ecosystem services, and (c) to approach stakeholders. Abbreviations: (a) Id for Identify; Pref for Preferences, (b) MCDA for Multi-Criteria Decision Aid. Note that in (a) asterisk includes only cultural services; in (b) community mapping includes only those studies using this technique to identify ecosystem services; therefore, the percentage of published articles about mapping ecosystem services might be much greater; outcomes refers to both focus groups and workshop results; economic refers to the percentage of articles using economic methods as a social valuation of ecosystem services (i.e., calculated separately from the other percentages); in (c) focus group also includes workshops.

Case study: The social assessment of ecosystem services in the River Piedra floodplain

Identifying ecosystem services and flows

Stakeholders perceived a general increase in ecosystem services over the last 50 years, mainly through cultural services such as recreation, tourism, and relaxation & life quality (Fig. 5). They also perceived a decrease in water-dependent services such as water quality regulation, energy generation (hydropower), leisure (swimming in the river), traditional ecological knowledge, raw material collection, food provision (fish and crabs), and local varieties (genetic resources) from upstream to downstream. The change in ecosystem services was perceived across stakeholder groups, indicating that changes affected all social groups considered. Additionally, interviewees pointed out valuable aspects of the ecosystem that are not usually included as ecosystem services: biodiversity, nature tourism (which provides job opportunities), traditional ecological knowledge, and health (such as disease prevention).

Ranking ecosystem services

Water supply, water quality regulation, and water flow regulation were the ecosystem services that were ranked the highest, whereas energy supply, raw material production, and medicinal plants were ranked the lowest. Responses within each stakeholder group varied, which prevented us from defining stakeholder groups according to their preferences for ecosystem services.

Discussion

In this paper, we go a step further in the social evaluation of ecosystem services by identifying three basic aspects that should be explicit in such assessments: (1) the spatial and temporal context (boundary delimitation); (2) the social context (who evaluates); and (3) the methodology used (how ecosystem services are evaluated). We aim to launch social valuations of ecosystem services not only as an isolated exercise in valuation, or restricted to merely valuate cultural ecosystem services, but to advance our knowledge on the value that society gives to ecosystems, to enable comparisons across studies, and to improve land management plans.

Although we tested the framework in a single case study (Felipe-Lucia 2012), our experience in socio-ecological research (Comín et al. 2005; Escalera-Reyes 2011), the insights gained from the literature review, and the fact that the outlines proposed are broad, enable us to propose this framework as a useful approach to guide the social assessment of ecosystem services in a wide context. Therefore, we encourage both researchers and practitioners to use this framework in other case studies to test its validity and to enhance it if any pitfalls are found.

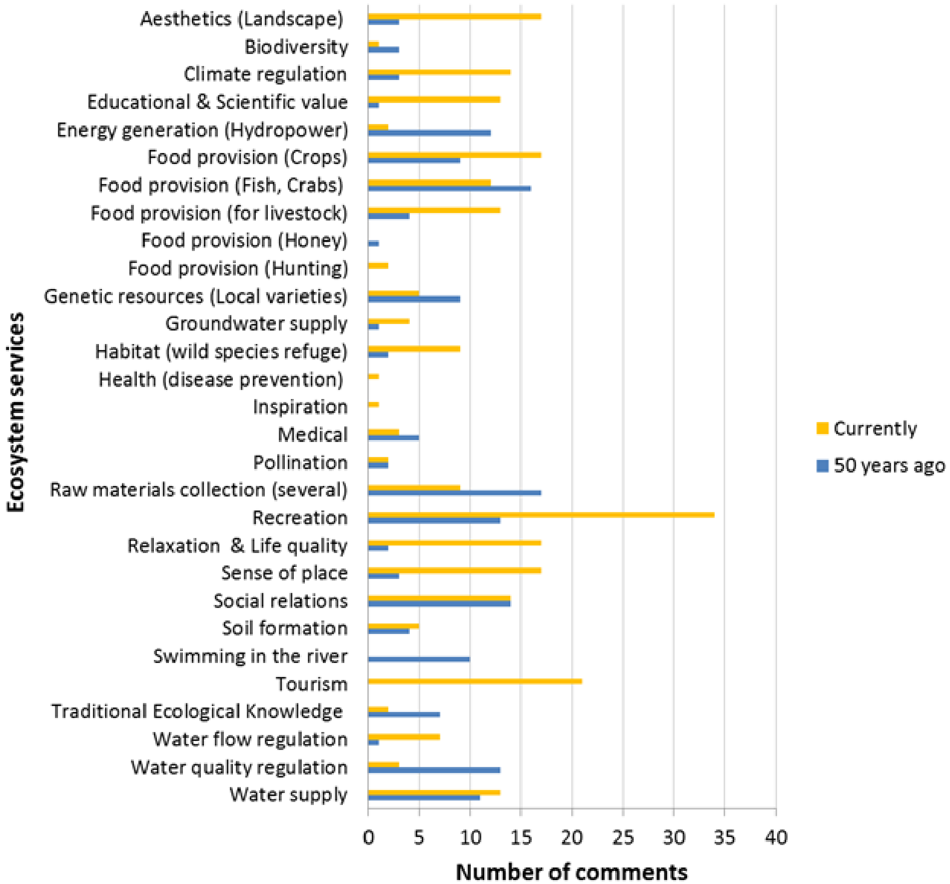


Figure 5. Identifying ecosystem services and flows in our case study. The Y axis represents the ecosystem services mentioned by stakeholders. The X axis represents the number of comments referring to each ecosystem service currently delivered in the study area (*yellow bars*) and 50 years ago (*blue bars*).

The review showed the potential of the social approach for ecosystems management, and also revealed some gaps in meeting such a challenge. At the stage of the spatial–temporal context, there are currently a low number of ecosystem services evaluations that gathered information across several spatial and temporal scales. However, considering such information would allow the flows of ecosystem services to be estimated. Combining both spatial and temporal flows can be useful to forecast future trends on the extent and direction of ecosystem services derived from land-use and land-cover changes over time (MEA 2005). Iteratively assessing social perceptions would predict support or tensions in society derived from the management actions accomplishing such changes (see Fig. 1).

Additionally, our review disclosed that the use of the social approach in the valuation of ecosystem services operated with the same type of ecosystems as studies from other approaches (Feld et al. 2009; Martin et al. 2012), and that there were some ecosystems not addressed at all. For instance, there is not much knowledge concerning the ecosystem services perception of the inhabitants of polar and desert ecosystems. This indicates that our understanding of the social value of ecosystem services across cultures can be expanded. Accounting with such information could expand our current perception of valuable ecosystem services and enhance management projects in remote areas.

Consistent with the results on the spatial context (the main spatial scales addressed were the supra-local and the municipality), the results of the social context showed that local residents were the group most frequently considered in the studies reviewed, but they were still in the minority among the studies. Listening to the local stakeholders and including their views and concerns might help the projects succeed. Even in larger projects, where decisions are made at the national or regional levels, implementing the views of representatives of local stakeholders whose well-being is affected is recommended (Hicks et al. 2009; Moreno et al. 2014). Neglecting local perceptions can hamper success in management projects that aim to enhance ecosystem services not supported locally (Hauck et al. 2013). Furthermore, on the other hand, projects or demands that arise at the local level are more likely to be implemented if they involve managers at the decision-making level, which are usually larger than local ones.

Regarding the suitability of the different methods exposed, we agree with Tallis et al. (2012) and Ringold et al. (2013) who suggest that an open combination of the two scopes identified would provide the most information, firstly identifying the valuable ecosystem services to stakeholders, and secondly, ranking their preferences (i.e., the value). This is especially important in land management, where trade-offs between alternative land uses are frequent, and a selection of ecosystem services to be enhanced or decreased might be required (Hicks et al. 2013).

In addition, we stress the need to clearly distinguish the social valuation from the economic valuation based on social preferences as separate approaches for the assessment of the ecosystem services. Our review showed that 264 papers outlined as “social valuations” were actually based on preferences revealed through methods using only monetary terms. Forty-five percent of our total records considered an economic valuation of some sort, 24 % were based on revealed preferences (including contingent valuation and “willingness-to-pay/accept/give-time” surveys), 17 % used choice experiments or modeling, 11% stated preferences, and 2 % used cost-benefit analysis. Given that we did not search for the term “economics” in our review, these figures might not be definitive. We provide them merely to draw attention to the fact that a large number of papers included in "social valuation" of ecosystem services are actually economic valuations based exclusively on social preferences. As we do not aim to expand on the differences between both approaches or the risks of limiting research on social preferences to monetary terms, we refer to other authors for further discussion (Funtowicz and Ravetz 1994; Chee 2004; Wegner and Pascual 2011; Farley 2012; Casado-Arzuaga et al. 2013). Defining clear methods for the social valuation of ecosystem services would strengthen the social approach as the alternative to economics to assess ecosystem services by society.

Finally, although in this paper we have developed one of the three approaches for the evaluation of ecosystem services, we understand that the three approaches together are required to properly assess the value of ecosystem services (Daily 1997) and to inform decision-making. In the example provided in Fig. 6, each ecosystem service (for example, clean water and fishing) is ascribed to more than one category–among them, regulating, supporting, provisioning, or cultural–as proposed by some authors (e.g., Chan et al. 2012a), and is evaluated using different indicators according to the approach adopted. Currently, most assessments intend to capture the whole value of ecosystem services by focusing solely on the ecological and economic approaches (Satz et al. 2013), while ignoring the social one (e.g., Kremen and Ostfeld 2005; Spangenberg and Settele 2010; but see Oteros-Rozas et al. 2012; Martín-López et al. 2014). Researchers probably assume that valuable ecosystem services are obvious and that they are able to identify them without including the opinion of society (Chan et al. 2012a), and even question whether using all three approaches might provide redundant measures (Brown 2013). However, it has been argued that using an integrated approach is the best way to make informed decisions based on ecological sustainability, economic efficiency, and social justice (Costanza 2000; MEA 2005; Farley 2012).

Practical recommendations:

We encourage scientists and practitioners to: (1) understand ecosystem services flows by comparing ecosystem services preferences across time and space, for which interviewers must clearly specify the temporal and spatial framework; (2) include a variety of stakeholders from all social ranges, grouping them according to their social characteristics and their use of the ecosystem; and (3) evaluate ecosystem services via both identification and ranking, insisting that stakeholders propose ecosystem services that are valuable to them, without listing constraints. For this third recommendation, ecosystem services need to be clearly defined, by indicating or separately evaluating the different benefits each ecosystem service can provide (Reyers et al. 2013). Also, the role of stakeholder representatives should be stated to ensure that they express the preferences of the organization they represent; such organizations should establish their own ranking of ecosystem services preferences.

Thus, we need to distinguish (i) the cultural services from the social approach and (ii) the social approach from the economic valuation based on social preferences. Additionally, we suggest taking the proposed framework into an iterative process, which deepens and evolves as do changes in the social–ecological context, human needs, and land uses.

Finally, the baseline question of whether we are actually able to establish our preferences for ecosystem services remains unsolved. In our western-culture society, we are so rarely asked to appreciate what we obtain for free and to put into practice our system of values that it is difficult for us to establish preferences for ecosystem services or even to identify the ecosystem services we receive. We believe that the underlying challenge of our society is to enable citizens to express their opinions for decision-making. Fair social participation in decision-making based on ecosystem services assessments leads to our well-being.

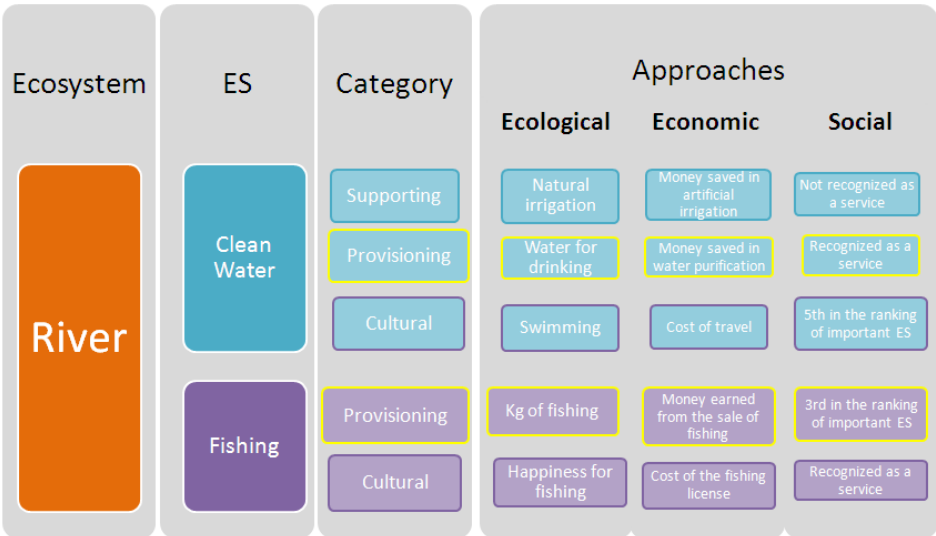


Figure 6. Example of approaches that can be applied to evaluate two ecosystem services (ES) provided by riverine ecosystems. Although each service is often ascribed to a unique category (*second column*, *blue box* for supporting and *purple box* for cultural), it can actually be evaluated by more than one category (*third column*, *blue frame* for supporting, *yellow frame* for provisioning, and *purple frame* for cultural). Furthermore, each category can be evaluated from the ecological, economic, or social approach, using different indicators. The assessment of all three approaches is strongly recommended for a complete valuation of ecosystem services.

Conclusion

To complement the ecological and economic assessments of ecosystem services, a three-step framework for the social valuation of ecosystem services is proposed. This framework provides a useful tool to contrast outcomes across studies and to support land planning and management. We address important questions at each stage, such as considering spatial–temporal flows, including stakeholders from all social ranges, and using two complementary methods (both identification and ranking) to value ecosystem services. Additionally, we stress the need to differentiate (i) the cultural services from the social approach and (ii) the social approach from the economic valuation based on social preferences. Defining clear methods for the social valuation of ecosystem services would strengthen this approach as the alternative to economics to assess ecosystem services by society. We aim to launch the social valuation of ecosystem services as a tool to enable citizens to express their opinions regarding decision-making. A fair social participation in decision-making based on ecosystem services assessments is the way to human well-being.

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[1] Note that in this case, *regional* refers to representatives from a county or province.

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S2. List of cards used to rank ecosystem services preferences by stakeholders.

Ecosystem services (left column) were valued by stakeholders through the ranking of preferences via cards (brightly coloured, originally in Spanish); these comprised one column with the topic, a second column explaining the main idea that each ecosystem services represents and a third column including an example that applies to the River Piedra floodplain. Here, we attempted to capture the widest range of ecosystem services valuable to stakeholders based on a previous observation of the main activities developed around the River Piedra floodplain.

Ecosystem Service	Topic	Main idea	Applied example
Climate regulation	**CLIMATE**	TEMPERATURE and environmental MOISTURE regulation	We feel much cooler under a poplar tree or by the river than in the middle of a field.
Water flow regulation	**WATER**	AQUIFER recharge, water drains underground	A part of the floodwater can be drained through the soil.
Water quality regulation	**WATER**	Water PURIFICATION: the stream water is purified downstream.	Turbidity disappears. Stream water is drinkable.
Water supply	**WATER**	Householdand farming CONSUMPTION.	Stream water is used for drinking, washing, cooking, gardening, and farming. Household water comes from a well by the side of the river.
Soil formation & Nutrient regulation	**SOILS**	Nutrient-rich FERTILE SOIL is created. Soils are retained.	Floods bring sediments that are good for farming. Riverbank vegetation prevents soil losses by floods.
Pollination	**PLANTS and ANIMALS**	POLINATION (in fruit groves and elsewhere).	Fruit-groves give fruits thanks to pollen carried by bees.
Genetic resources	**PLANTS and ANIMALS**	LOCAL VARIETIES of vegetables, herbs and animals	Our local varieties better support low temperatures and droughts. In the river there are animals only found in this area.
Medicinal	**PLANTS and ANIMALS**	MEDICINAL	We use plants for remedies.

Ecosystem Service	Topic	Main idea	Applied example
Food provision (Crops)	**PRODUCTION**	CROPS (orchards, fruit groves, cereal) for selling or own consumption	Crops generate richness.
Food provision (Fish, Crabs, Hunting)	**PRODUCTION**	FISHING, HUNTING	Fishing and hunting are good to obtain food.
Sheep Food provision	**PRODUCTION**	PASTURES for livestock	Livestock use pastures or stubble by the river.
Raw materials	**PRODUCTION**	MATERIALS, wood, etc.	Woods and gravels are used for building.
Energy	**ENERGY**	HYDROELECTRICITY (la Requijada, la Tranquera)	Hydroelectricity is generated by the river
Landscape & Aesthetic & Habitat provision	**LANDSCAPE**	A VARIETY of LANDSCAPES	The river provides the landscape with a different value.
Sense of place & Social relations	**CULTURAL**	SENSE-OF-PLACE SOCIAL RELATIONSHIPS	I feel attached to this place because of the river. There is a festival we celebrate together by the river.
Recreation & Tourism	**CULTURAL**	LEISURE TOURISM	We usually walk the river or fish there. Tourism attracts money here.
Educational & Scientific value	**CULTURAL**	EDUCATIONAL	School-children come to the river to learn.
Spiritual & Relaxation & Life quality	**CULTURAL**	RELAXATION LIFE QUALITY	The river is relaxing. Living close the river or being able to come to visit provides me with a good life quality.

CAPÍTULO 6. ECOSYSTEM SERVICES FLOWS: WHY STAKEHOLDERS' POWER RELATIONSHIPS MATTER*

ABSTRACT. The ecosystem services framework has enabled the broader public to acknowledge the benefits nature provides to people. However, not all people benefit equally from these services. Rather, power relationships are a key factor influencing the access of individuals or groups to ecosystem services. In this paper, we propose an adaptation of the 'cascade' framework for ecosystem services to integrate the analysis of (1) ecological interactions among ecosystem services and (2) stakeholders' interactions, reflecting power relationships that mediate ecosystem services flows. We illustrate its application using the floodplain of the River Piedra (Spain) as a case study. Our analyses were useful to detect: (i) keystone ecosystem services that determine the provision of other ecosystem services, (ii) relevant services for each stakeholder group, (iii) the ability of stakeholders for managing each service and their implications in other ecosystem services, and (iv) power asymmetries between stakeholders derived from their capacity for managing ecosystem services. With these analyses, we identified the 'formal' power relationships exerted by stakeholders according to their ability to access and manage ecosystem services, and the mechanisms they use to exert power. Our results revealed that the strongest power was held by those stakeholders who managed (although did not use) those keystone ecosystem properties and services that determine the provision of other services (i.e., intermediate regulating and final services). In contrast, non-empowered stakeholders were only able to access the remaining non-excludable and non-rival ecosystem services (i.e., some of the cultural services, freshwater supply, water quality, and biological control). Finally, we discuss the implications of uncovering power relationships that mediate access to ecosystem services for the management of ecosystem services flows and social-ecological systems.

Key words: ecosystem services beneficiaries; environmental management; inequality; power asymmetries; social-ecological interactions; trade-offs.

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Introduction

The ecosystem services framework [1] has enabled the broader public to acknowledge the benefits nature provides to people [2]. These include tangible or material benefits such as provisioning services (e.g., food, raw materials) and intangible or immaterial benefits such as cultural services (e.g., recreation, relaxation, environmental education, and aesthetic enjoyment), regulating services (e.g., nutrient regulation and climate regulation), and supporting ecosystem properties (i.e., the underlying mechanisms of the ecosystems) such as habitat provision and soil formation. However, not all people benefit equally from these ecosystem services. Recent research highlighted spatial characteristics as drivers of inequalities in ecosystem services provision [3,4]. For example, whereas ‘upstream’ populations may benefit from water quality, ‘downstream’ populations may not. Yet, the potential of ecosystems to benefit people not only depends on the spatial characteristics of the flow of services [5–8] but are derived from their multiple types of interactions [9]. On the one hand, these depend on the inner interactions among the ecological components of the ecosystems [10] and on the interactions among ecosystem services causing trade-offs and synergies [11]. On the other hand, the interactions among stakeholders, which are partially caused by power relationships, can determine the access to and management of ecosystem services. Power relationships are a well-known concept used in natural resource management to determine asymmetries in the access to resources [12–17]. Power relationships are also well-known in social sciences and are used to uncover the consubstantial asymmetries in social relations [18–22]. For instance, ecological anthropology and political ecology already incorporate the concept of power to human-environment interactions [23]. In ecosystem services literature, studies analysing power relationships are developed in the context of payments for ecosystem services [24,25], although power relationships can modulate either the stakeholders’ use of ecosystem services or the interactions between the ecosystem services supplied. Power asymmetries among stakeholders mean that some stakeholders may use a particular ecosystem service or a set of ecosystem services while other stakeholders might be excluded. Therefore, power asymmetries can create social conflict [4,26], and affect stakeholders’ well-being [27]. For instance, empowered stakeholders can decide about the ecosystem services supplied and regulate access to them, negatively affecting non-empowered stakeholders by reducing their ability to access ecosystem services. In addition, management decisions ultimately driven by power relationships modulate ecosystem services interactions resulting in trade-offs between ecosystem services [9,28]. Therefore, power relationships emerge as a key factor influencing: (i) people’s access to ecosystem services; (ii) stakeholders’ interactions and roles regarding ecosystem services; and (iii) environmental management shaping the provision of ecosystem services.

Including the concept of power relationships into ecosystem services research enables to expose the gap between the production of services by an ecosystem and the actual benefits people receive. Such gaps can reveal those people dependent on certain ecosystem services for their well-being that are at risk of being excluded from accessing ecosystem services [27]. Moreover, power relationships, including the beneficiaries of ecosystem services, the contributors to services production, and those who are excluded (i.e., the losers [29]) have not yet been integrated into ecosystem services management [30]. Hence, integrating power relationships into ecosystem services research emerges as a key challenge as, to our knowledge, no study has formalized nor empirically analysed this component that mediates ecosystem services flows.

In this context, the general aim of this study was to reveal the role of power relationships for ecosystem services flows from the supply by the ecosystems to the users. In order to address this general aim, in the next section we describe the adaptation of the ecosystem services 'cascade' framework to integrate the analysis of ecological interactions among ecosystem services and of power asymmetries among stakeholders that determine the use and management of ecosystem services. Then we describe the methods used to apply the conceptual framework to the River Piedra case study (NE Spain). The results section shows the main findings related to the dependence relationships among the ecosystem services analysed and the role of stakeholders mediating access to ecosystem services through the identification of power asymmetries. In the discussion section we address the applicability of the conceptual framework and the implications for accessing ecosystem services of both power imbalances among stakeholders and the excludable and rival characteristics of ecosystem services. Finally, we provide some insights for environmental management to deal with social-ecological interactions along the flow of ecosystem services.

Conceptual framework

The 'cascade' framework depicts ecosystem services as a flow from the ecosystem towards human well-being [31]. This framework has been gradually modified to incorporate ongoing developments of ecosystem service science [32–34], such as the introduction of societal processes in the step from 'service' to 'benefit' [35]. We propose to further refine this step by identifying both the interactions among ecosystem services and among stakeholders that mediate and could impair people's access to ecosystem services (Fig. 1).

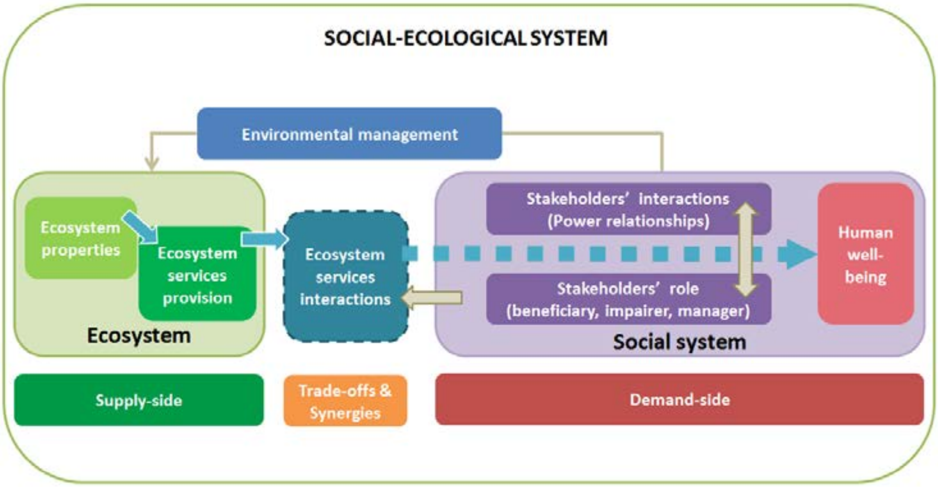

Figure 1. Conceptual framework of the interactions along the flow of ecosystem services from the supply-side to the demand-side and human well-being. The framework highlights both the interactions among ecosystem services and among stakeholders that mediate and could impair people's access to ecosystem services. Blue arrows represent the flow of ecosystem services. Beige arrows denote interactions within or from the social system (Inspired from Haines-Young and Potschin [31], Martín-López et al. [33], Spangenberg et al. [35]).

Ecosystem properties (i.e., the biophysical structure and functioning of ecosystems) contribute to provide ecosystem services and human well-being. However, ecosystem services are not isolated independent units, but rather depend on each other [36] and interact causing trade-offs and synergies [11] (see Box 1 for definitions). The flow of ecosystem services is shaped through the social system by several types of complex interactions among multiple stakeholders. First, stakeholders interact among themselves through different types of relationships that are modulated by formal power asymmetries (e.g., property rights, access, or legal permissions), informal power asymmetries (e.g., social leadership, gender inequity), or hidden power imbalances (e.g., social pressure promoting self-censorship). Second, stakeholders play different roles in the ecosystem according to their type of relation with each ecosystem service. Basically, they can manage ecosystem services (i.e., co-producing or impairing them), or be recipients of ecosystem services (i.e., using them but also being excluded from access) [30], although a single stakeholder could perform several of these roles [37]. Stakeholders' roles depend on their preferences towards ecosystem services, which in turn might differ across stakeholders according to their needs, values, and power asymmetries [38–42]. In addition, stakeholders' interactions affect the role of individual stakeholders in the system, which in turn perpetuates their power relationships. Last but not least, the social system (i.e., stakeholders' interactions, roles, and preferences) drives environmental management, establishing the management and use of ecosystem services and conditioning the ecosystem properties responsible for ecosystem service provision [43,44]. Additionally, the use of ecosystem services by stakeholders can entail trade-offs and synergies among ecosystem services [9,28,45].

Box 1. Key concepts related to ecosystem services and definitions. Concepts are listed according to the order they appear in the text.

Concept	Definition
Trade-off	Situation in which land use or management actions increase the provision of one ecosystem service and decrease the provision of another. This may be caused by simultaneous responses to the same driver or caused by true interactions among ecosystem services (adapted from [11]).
Synergy	A win-win situation that involves a mutual improvement of two ecosystem services (adapted from [45]).
Stakeholder	Any group, organization or individual having a stake, interest, or who can affect a biological or physical resource, ecosystem service, institution or social system, or someone who is or may be affected by a public policy (adapted from [29], [37]).
Power relationships	The human ability to control or influence the access of others to ecosystem services.
Beneficiary	Stakeholders who directly use and benefit from ecosystem services [37].
Impairer	Stakeholders who negatively affect the provision of ecosystem services as a consequence of their direct or indirect use (adapted from [37]).
Manager	Stakeholders who directly influence the way ecosystem services are provided or can be used [37].

Methods

Study area

The study area comprises the municipalities across the River Piedra (616 km^2) in NE Spain (Fig. 2), which is characterized by marked season variability in the water flow. The upper part of the River Piedra (ca. 46 km) is dry for most of the year due to a combination of a semiarid climate and a calcareous substrate. The lower part of the river is permanent (ca. 30 km), as it receives the groundwater discharge from the upper basin. River flow rates in most of the lower reaches are usually inverted to natural rates, as the river is retained in a 78.8 hm3 reservoir whose regulation depends on the water demand from irrigators from adjacent basins. The floodplain is characterized by agricultural use (46.6%), including dry cereal crops in the upper lands, irrigated cereal crops and poplar groves in the central part, and fruit groves and orchards in the lower lands. Natural areas (2.5%) are restricted to the upland gorges (usually dry) located between the municipalities of Aldehuela de Liestos and Embid, and to a private natural park, the Monasterio de Piedra, located in the municipality of Nuévalos. The park's main attraction is the large number of waterfalls of the River Piedra, which contrast hugely with the semiarid surrounding landscape. The tourism generated by the park is the main economic driver of the area, and attracts tourists to other nearby amenities and activities (e.g., restaurants, lodges, trekking, mountain-biking, ornithology, fishing, kayaking). The population is weakly structured, dominated by elderly people and significantly more men than women, although this trend reverses during school holidays.

Data collection

<u>Ecosystem services supply</u>

We identified the key ecosystem services provided by floodplains following Harrison et al. [46] at European scale, and Vidal-Abarca and Suárez [47] at national scale, as well as from prior knowledge of the functioning and the ecosystem services of the study area [28]. We gathered available data of 12 ecosystem services that were relevant to maintain the flow of services in the area. The selection included two supporting ecosystem properties (soil conditions, composed of soil formation and soil stability, and habitat quality), four regulating services (nutrient regulation, carbon sequestration, biological control, and water quality), three provisioning services (freshwater supply, food production, and raw materials), and three cultural services (aesthetic, recreation, and environmental education). Table 1 synthesizes the methods used for measuring each ecosystem service; for further details about methods see S1 in Supplementary Information (SI).

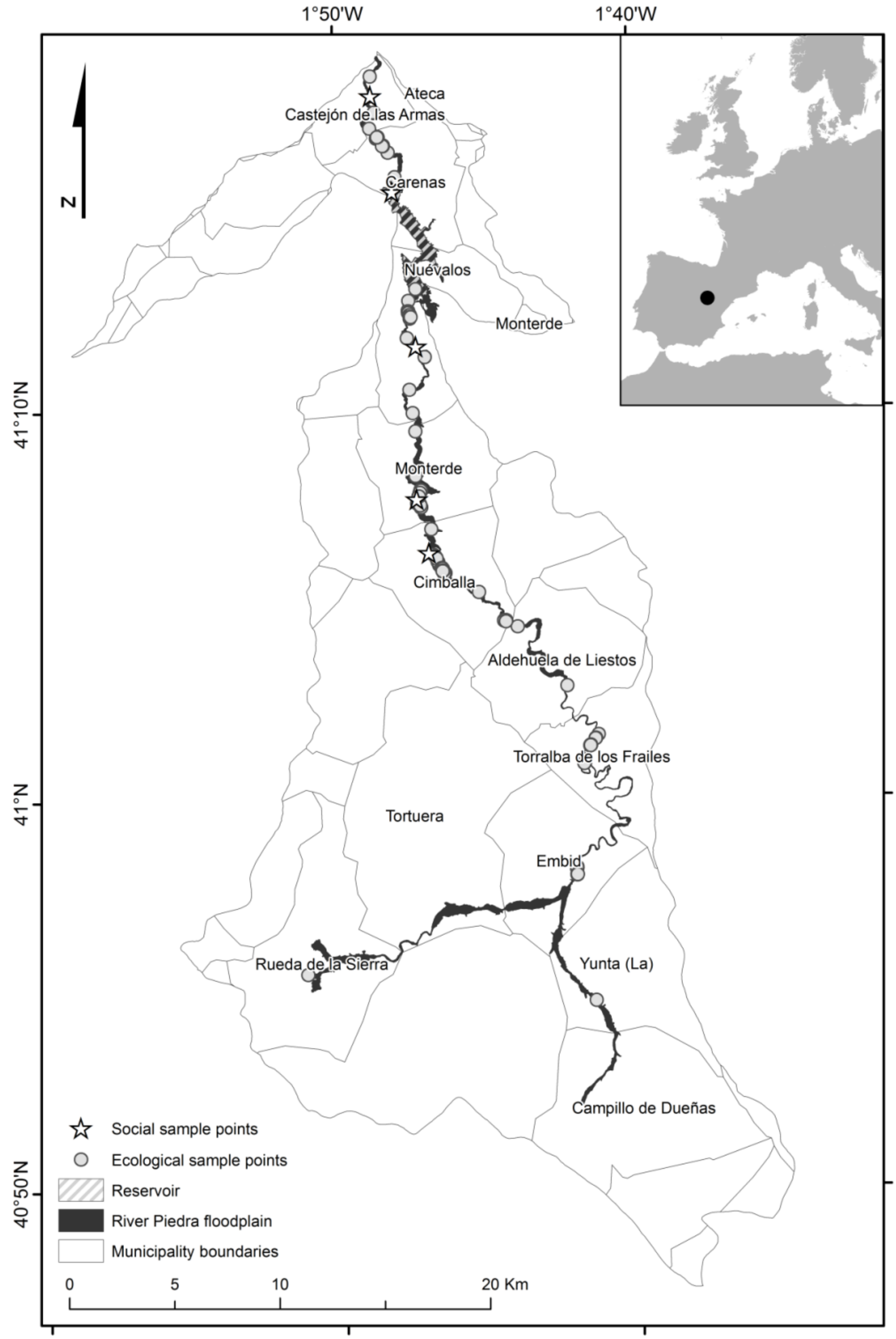

Figure 2. The watershed of the River Piedra in NE Spain divided by municipality boundaries. Dots indicate ecological sample points and stars social sample points (note that external stakeholders are not represented in this figure).

Table 1. Description of the supporting ecosystem properties and ecosystem services identified.

Category	Type	Name	Indicator	Method (units)	Number of sample points	Years collecting data	Reference
Supporting	Intermediate	Habitat quality	RQI	Riparian Quality Index (unitless)	21	2011, 2012	[48]
Supporting	Intermediate	Soil conditions	Soil formation	Organic matter content in top soil (percentage)	324	2011, 2012	[49]
			Soil stability	Organic matter layer in top soil (cm)	324	2011, 2012	
Regulating	Intermediate	Water quality	NO_2^-	Nitrite content in water (ppm)	281	2009- 2011	[50]
			NO_3^-	Nitrate content in water (ppm)	281	2009-2011	[50]
			PO_4^-	Phosphate content in water (ppm)	281	2009-2011	[50]
Regulating	Intermediate	Nutrient regulation	C	Total carbon in top soil (percentage)	324	2011, 2012	
			N	Total nitrogen in top soil (percentage)	324	2011, 2012	
			P	Total phosphorus in top soil (percentage)	324	2011, 2012	
Regulating	Intermediate	Biological control	Plant strata	Number of plant strata (number)	54	2011-2013	[51]
Regulating	Final	Carbon sequestration	CO_2 sequestration	CO_2 equivalent tons sequestered by plants (CO_2 eq tons/year)	21	2011	[52]

Provisioning	Final	Freshwater supply	Water consumption	Cubic meters of water concessions per municipality (m^3/year)	12	2011	[53]
Provisioning	Final	Food production	Yield	Kilograms per hectare (kg/Ha · year)	21	2011	[54]
			Calories	Kilocalories per hectare (kcal/Ha · year)	21	2011	[54]
			Money	Euros per hectare (€/Ha · year)	21	2011	[55]
Provisioning	Final	Raw materials	Production	Tons of annual biomass increase (tons/year)	21	2011	[52]
Cultural	Final	Aesthetic	Pictures	Number of pictures uploaded to Panoramio (number)	84	2014	[56]
Cultural	Final	Recreation	Fishing	Meters of river available for fishing (m)	84	2012	[57]
			Sports	Extent of floodplain viewshed from open access trails (Ha)	84	2012	[58]
			Picnic areas	Number of designed picnic areas (number)	84	2012	[28]
Cultural	Final	Environmental education	Educative panels	Number of educative panels (number)	84	2012	[28]

Ecosystem services benefits

To learn about the ecosystem services used in the floodplain of the River Piedra and the limitations to benefiting from these services, we conducted 71 face-to-face, semi-structured interviews with the main stakeholders of the study area. These included residents, holidaymakers, farmers, tour operators (hosting or guiding nature tourists), local mayors, local teachers, scientists, nature protection agents, and technicians working on riverbank restoration projects. These people were related to municipalities within the permanent river flow and their daily life was strongly related to, or dependent on the riparian ecosystem. Therefore, the municipalities within the seasonal river flow were excluded from this study as they do not perceive themselves to live within a riparian ecosystem and their activities depend little on this ecosystem. The targeted local population comprised 880 inhabitants [59] from five municipalities (see Fig. 2). Table 2 presents a classification of stakeholders and their brief description. Interviewees were asked about the status of the riparian ecosystem, the causes and solutions to solve the problems identified, and about the uses, products, and benefits they derived from the valley of the River Piedra. A minimum number of ten people from each of the main stakeholders' groups were interviewed until the information received was saturated (i.e., we did not receive any new information from the same sector of stakeholders [60]). Interviews were performed by the first author between August 2011 and March 2012 and lasted between 30 and 90 min. Digital records of the interviews were made with the interviewees' consent. Interviews were transcribed and coded for further analysis (see Table S2.1 in SI for details of the interviewees).

Table 2. Stakeholders' groups, names, number of respondents, and description.

Group	Name	n	Description
1	Primary sector	16	Farmers (including both land owners and land tenants of orchards, fruit groves, irrigated and dry cereal crops, and poplar groves), shepherds, and workers at a fish farm.
2	Recreation sector	13	Owners or workers at restaurants, hotels, lodges, nature tour operators, adventure enterprises, and at the Monasterio de Piedra (i.e., a regional touristic site).
3	Leisure	26	Retired residents, visitors, hikers, bikers, fishermen, etc.
4	Institutions	16	Local councils. Government bodies: the regional water management body (Confederación Hidrográfica del Ebro), which depends on the Ministry of the Environment; Nature Protection Agents, which depend on the regional government (Gobierno de Aragón). Scientific and educational institutions: scientists from the Pyrenean Institute of Ecology (IPE – CSIC) and the University of Zaragoza; teachers from the local elementary school and high school. Technicians from a public company working on environmental projects on the riverbanks and the floodplain of the River Piedra.

Data analysis

Ecosystem services supply

To model the flow of ecosystem services, we built an initial path model (Fig. 3) on the basis of the classification of ecosystem services as intermediate or final. To build the path model, we performed an expert panel in May 2014 composed of four experts from the fields of ecosystem service science, conservation ecology, and limnology that independently modelled the flow of ecosystem services in the study area. Structural equation modelling (SEM) is a statistical technique to model complex multivariable relationships among observed and latent variables, which includes two models: the relations between the manifest (observed) variables and their own latent variable, and the relations among all latent variables (exogenous or endogenous) [61]. In our model, supporting ecosystem properties were considered as exogenous variables (i.e., independent from other ecosystem services), and intermediate regulating services and final services as endogenous variables (i.e., dependent on other ecosystem services) (Fig. 3). Thus, in the River Piedra case study, soil conditions and habitat quality were the supporting ecosystem properties from which ecosystem services depended directly (linked by an arrow) or indirectly (linked through an intermediate regulating service). Nutrient regulation is a regulating service which directly depended on soil conditions and habitat quality. Carbon sequestration depended on habitat quality (because the quantity of trees mediates carbon sequestration) and nutrient regulation (also mediates trees' performance). Biological control depended on habitat quality as the number of plant strata hosting species performing different functions in the ecosystem depends on it. Water quality was related to habitat quality (e.g., a good quality of riparian habitats avoids runoffs into water) and to nutrient regulation (e.g., through regulating nitrogen and phosphorus content in soils). Freshwater supply was connected to both habitat and water quality as water supplied needs to be in a good ecological and chemical status which is mediated by a good habitat quality. Food and raw materials production was related to freshwater supply (e.g., increasing water for irrigation increases productivity) and to regulating services (nutrient regulation and biological control), whereas cultural services were directly linked to habitat quality.

We followed a formative SEM approach [62], in which each latent variable is related to its manifest variable by a linear function plus a residual term. We normalized all manifest variables to ensure homogeneous weights and checked the unidimensionality of the blocks of manifest variables using the criteria: i) Cronbach's alpha higher than 0.7 [63] and ii) Dillon-Goldstein's rho higher than 0.7 [64]. Manifest variables not matching these criteria were dropped from the initial model. The quality of the final model was assessed using: i) the Goodness of Fit index [65]; ii) the adjusted

R2 of the latent variables; and iii) the average communalities [66]. All statistical analyses were performed with the software XLSTAT (2014.3.01).

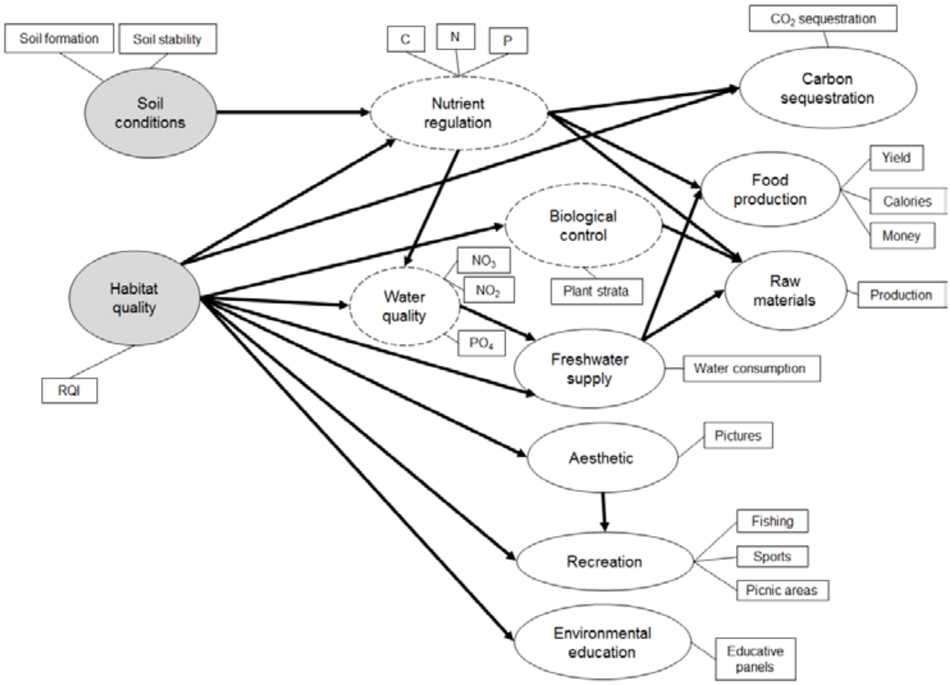

Figure 3. Conceptual diagram of the initial structural equation model (SEM) based on those paths among variables determined by the expert panel. Latent variables (i.e., ecosystem properties or services) are denoted by ellipses, while manifest variables (i.e., the indicators used) are inside a box. Supporting ecosystem properties (i.e., exogenous variables) are shaded, intermediate regulating services are dashed, and final services are solid.

Ecosystem services benefits

The ecosystem services mentioned by each stakeholder group during the interviews (see Table S2.2 in SI) provided evidence of their role in relation to ecosystem services. According to this information, we linked each stakeholder group to the services they used, contributed to produce, impaired, or managed. Additionally, we classified the ecosystem services identified within a gradient from rival to non-rival, and from excludable to non-excludable through an expert panel. The panel was held in June 2014 and comprised six experts from the fields of ecosystem services, policy, and land management that independently classified ecosystem services according to the characteristics of this case study. In cases of divergence, the moderator of the panel unified the classifications according to the comments provided by each expert. We followed the approach of Costanza [67] (p. 351) which defined rival ecosystem services as those that can be consumed ("the degree that one person's benefiting from them interferes with or is rival with other's benefiting from them"), and excludable ecosystem services as those that can be privatized ("the degree that individuals can be excluded from benefiting from them"), and we incorporated the concept of congestible (i.e., moving from non-rival to rival if excessive use decreases their good initial conditions) suggested by Fisher et al. [3]. We used this classification to represent each ecosystem service in a diagram showing stakeholders' use versus ability to manage ecosystem services by adapting the approach proposed by Reed et al. [16] and Iniesta-Arandia et al. [68], where the former identified four clusters according to the degree of power and interest of stakeholders and the latter according to their degree of dependence and influence. Additionally, we included a variant of this diagram displaying stakeholders' use versus their ability to impair ecosystem services.

Ethics statement

Part of these analyses is based on interviews results. The interviewees were voluntary, and their answers were confidential and anonymized for analysis. Participants verbally consented to participate in this study under these conditions. Written consent was not requested in order to facilitate the interactions between interviewer and participants. We cannot document participant consent because we only started recording once participants had given their agreement. When participants did not agree to be recorded but consented to participate, written notes were taken by the interviewer. The Academic Commission of the Doctorate Program in Environment and Society of the Universidad Pablo de Olavide (Seville, Spain) approved this study and this consent procedure. Additionally, the Instituto Pirenaico de Ecología – CSIC, approved the methods used in field sampling.

Results

Dependence relationships among ecosystem services on the supply side

The results of the SEM highlighted the fact that some ecosystem services were strongly dependent on others, while others were less dependent (Fig. 4). Ecosystem properties (i.e., soil conditions and habitat quality) were key variables, given the significant effects they had on all ecosystem services to which they were related, except to freshwater supply. Soil conditions had a significant strong effect on nutrient regulation ($\beta = 0.758$). The largest effects of habitat quality ($\beta \geq 0.5$) were on environmental education, recreation, biological control, and carbon sequestration. Habitat quality also had significant but weak ($\beta \geq 0.1$) effects on water quality and aesthetics, and a weak significant negative effect on nutrient regulation (β = -0.135); this latter could be explained by the different types of land uses included in the assessment, for which there might be opposite relationships. For instance, perennial forests have excellent quality (as assessed using the Riparian Quality Index [48]) but do not contribute much to nutrient regulation; conversely, cultivated land uses might have high concentrations of nutrients due to human inputs through fertilization. Intermediate services also had significant but weaker effects on final ecosystem services: biological control had a positive effect on raw materials (β = 0.379) and nutrient regulation had a negative effect on food production (β = -0.180), which can be explained by the fact that increasing food production can reduce nutrient regulation. Final services also displayed some interactions, namely, aesthetics had a weak significant positive effect on recreation (β = 0.153).

The adjusted R2 values are the variance explained by the model, so higher values are related to the ecosystem services explaining the most variance, and thus, to important variables of the model. In our model, environmental education (R2a = 0.817), recreation (R2a = 0.784), and nutrient regulation (R2a = 0.488) were the ecosystem services explaining the most variance. Less variance (R2a < 0.5) was explained by carbon sequestration, biological control, and raw materials. The variables explaining the least variance (R2a ≤ 0.1) were water quality, freshwater supply, food production, and aesthetics (Table 3).

The contribution of each service is represented by arrow thickness in Fig. 4 and highlights the main interactions maintaining the flow of ecosystem services. In our case study, supporting ecosystem properties (soil conditions and habitat quality) strongly influenced intermediate regulating services and cultural services, indicating that these are the key variables that mediate the flow of ecosystem services.

Restrictions to use ecosystem services

Four stakeholders groups were identified (see Table 2): primary sector, recreation sector, leisure and institutions. The identification of the ecosystem services linked to each stakeholder group was useful to detect: (i) key ecosystem services for each stakeholder group in terms of their use; (ii) the ability of stakeholders to manage each service; and (iii) power asymmetries derived from the management of ecosystem services through the identification of those stakeholders who are unique managers of particular services (Fig. 5). The primary sector (Group 1) was linked to most ecosystem services, either by using, co-producing, or impairing them; in addition, they were the main managers of two provisioning services (raw materials and food production). The recreation sector (Group 2) used and impaired water-related services, and used and co-produced carbon sequestration and cultural services, part of which they had great ability to manage. Leisure (Group 3) was linked to cultural and water-related services and to biological control. These three groups were also indirectly linked to habitat quality. The ecosystem services linked to institutions (Group 4) were used indirectly, except environmental education, which was co-produced by a section of this group (i.e., the governments and scientists), and used by the other section (i.e., the schools and universities). Further, this group was the main manager of habitat quality, water quality, and freshwater supply.

These results together with a general overview of the power relationships among stakeholders enabled us to classify the ecosystem services of this case study within a rival/non-rival and excludable/non-excludable matrix (Table 4, see S3 in SI for details).

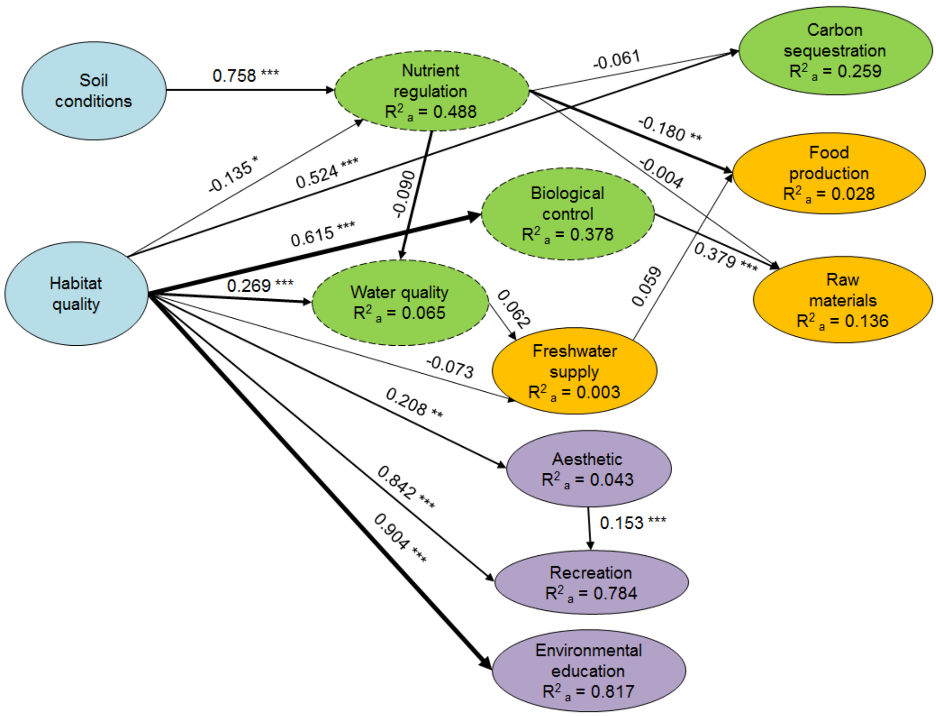

Figure 4. Structural equation model (SEM) results, showing the relationships between ecosystem services. Colours indicate the type of ecosystem service (green=regulating, gold=provisioning, purple=cultural) and supporting ecological properties (blue). Intermediate regulating services are dashed and final services are solid. Arrow thickness represents the percentage of the contribution to each service and numbers near arrows indicate the standardized regression coefficient. The asterisks denote significance (* p ≤ 0.05; ** p ≤ 0.01; *** p ≤ 0.001).

Table 3. Latent variables, adjusted R^2 (R^2_a), average communality (Ave. Com.), and Dillon-Goldstein's (D.G.) Rho from the structural equation modelling (SEM).

Latent variable	R^2_a	Ave. Com.	D.G. Rho
Habitat quality		1.000	1.000
Soil conditions		0.558	0.747
Nutrient regulation	0.488	0.316	0.679
Biological control	0.378	1.000	1.000
Water quality	0.065	0.592	0.823
Freshwater supply	0.003	1.000	1.000
Food production	0.028	0.606	0.858
Raw materials	0.136	1.000	1.000
Aesthetic	0.043	1.000	1.000
Recreation	0.784	0.639	0.799
Environmental education	0.817	1.000	1.000
Carbon sequestration	0.259	1.000	1.000
Relative Goodness of Fit (0.858)			

Table 4. Classification of the ecosystem services used in the River Piedra floodplain according to a rival/non-rival and excludable/non-excludable gradient.

	Excludable ↔	**Non-Excludable**
Rival *(High use)*	Food provision Raw materials Freshwater supply* Aesthetic* Recreation* Environmental education*	
Congestible ↕	Soil conditions* Nutrient regulation* Habitat quality*	Water quality Soil conditions* Nutrient regulation* Habitat quality*
Non-Rival *(Low use)*		Carbon sequestration Biological control Freshwater supply* Aesthetic* Recreation* Environmental education*

* Ecosystem services that can fall into several classifications according to specific situations. See main text for examples. Adapted from Fisher et al. [3].

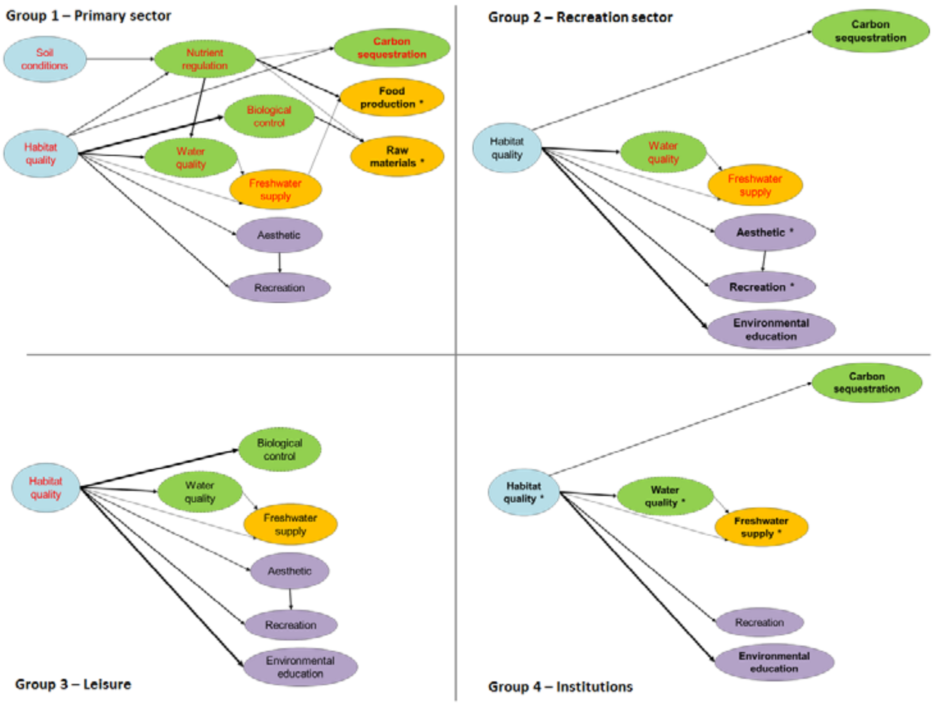

Figure 5. Ecosystem services related to each stakeholder group. Colours indicate the type of ecosystem services (green=regulating, gold=provisioning, purple=cultural) and supporting ecological properties (blue). Intermediate regulating services are dashed and final services are solid. Impaired ecosystem services are in red, ecosystem services managed or co-produced are in bold, and they are marked with an asterisk (*) when managed by a single group. Note that habitat quality and carbon sequestration were only indirectly used by groups 1, 2 and 3, and that all ecosystem services linked to group 4 (excluding environmental education) were used indirectly.

How stakeholders mediate access to ecosystem services

We related stakeholders' roles to the rival/excludable classification of ecosystem services by depicting each service in a diagram showing stakeholders' use versus their ability to manage ecosystem services (Fig. 6a) and stakeholders' use versus their ability to impair ecosystem services (Fig. 6b). The results on Fig. 6a highlighted the effect of power relationships on access to ecosystem services and differentiated five types of clusters, which mostly corresponded to the previous stakeholder classification. Fig. 6b provided complementary information especially useful to identify situations in which the same stakeholder group used and impaired the same ecosystem service.

In power relationships (Fig. 6a), the first cluster of stakeholders was those with a high ability to manage but low use of ecosystem services. This cluster was mainly composed of institutions and farmers (from the primary sector) being the only managers of key ecosystem services that determine the provision of other ecosystem services (Fig. 6a, top left corner). Within this cluster, the most dominant was group 4 (Institutions) because they mostly managed a key supporting ecosystem property (habitat quality) and key intermediate regulating services (water quality) able to maintain the ecosystem services flow and, thereby, affect other stakeholders. Further, they also managed final services (carbon sequestration, freshwater supply, and recreation) without using them. A secondary cluster here was farmers (Group 1: primary sector), as unique producers of food and raw materials (Fig. 6a, top left corner, dotted line); however, as these provisioning services are mostly exported outside the area, farmers just use them marginally (i.e., most of the production of such services is not on a self-consumption basis; rather, the income of these stakeholders mostly comes from the export of these goods).

The third cluster of stakeholders comprised those with a high ability to manage and to use ecosystem services. This cluster was mainly composed of group 2 (Recreation sector), which contributed to produce cultural services by offering aesthetic enjoyment, recreation, and environmental education, exclusively managed some of these services, and also used them, benefiting from the tourism generated (Fig. 6a, top right corner). Within this cluster, the contribution of group 4 (Institutions) to environmental education was split because scientists and the government provided the area with educative panels to explain ecosystem functioning, whereas schools and universities benefited from having such panels or directly benefited by learning from the ecosystem.

The next cluster of stakeholders comprised users of ecosystem services with low or no ability to manage them (Fig.6a, bottom right) and was composed of the leisure (Group 3) and primary sector (Group 1). Consequently, these stakeholders benefited from just the remaining non-excludable and non-rival ecosystem services (a part of cultural services, freshwater supply, water quality, and biological control). An intermediate cluster here was the primary sector (Group 1, Fig. 6a, right dotted line), who had some opportunities to manage the services they used the most (nutrient regulation, soil conditions, and freshwater supply) through their farming practices. However, this group impaired these same ecosystem services by overuse, driving non-rival ecosystem services to rival (Fig. 6b, dotted line). Still in the second diagram, we observed that in the case of biological control, the use of the service did not directly imply degradation; rather this service was impaired by the farming practices used, which in turn may potentially affect other stakeholders. Regarding carbon sequestration, a service of global extent, farming practices did not directly affect other

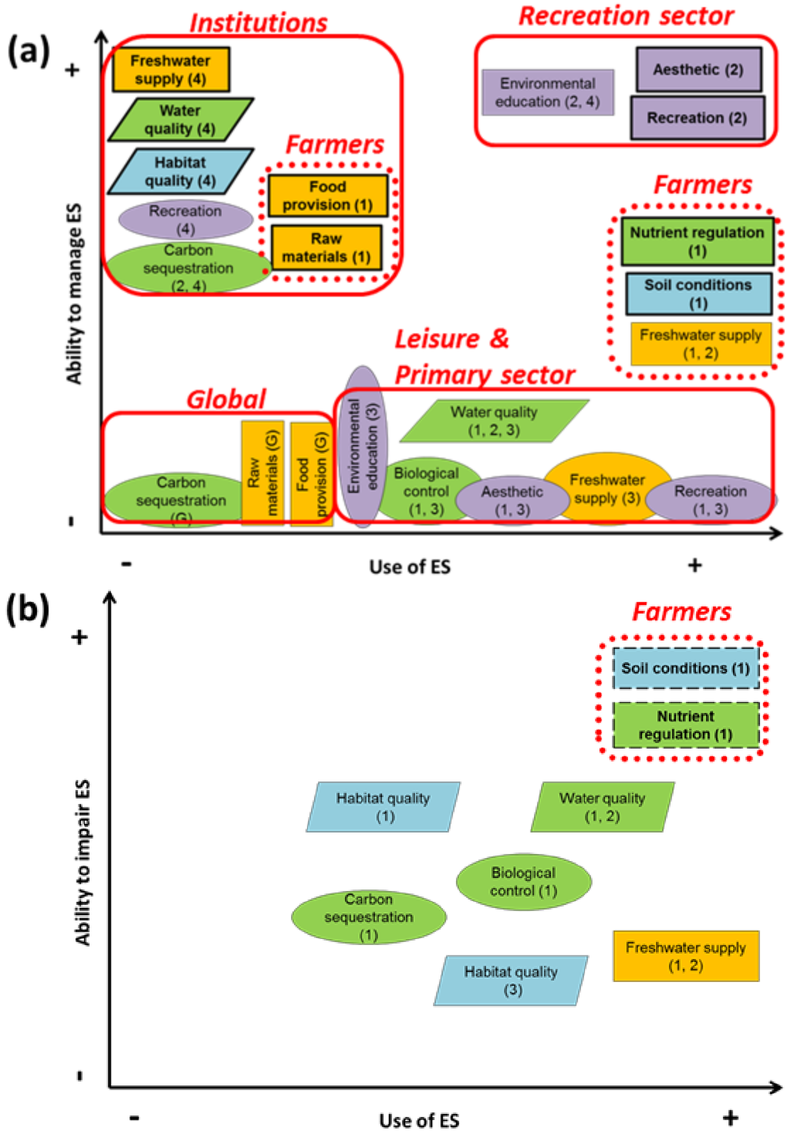

Figure 6. (a) Stakeholders' use versus ability to manage ecosystem services (ES), and (b) Stakeholders' use versus ability to impair ecosystem services (ES). The colour of the box indicates the type of ecosystem service (green=regulating, gold=provisioning, purple=cultural) and supporting ecological properties (blue). Rival and excludable services are in rectangles, non-rival and non-excludable services are in ellipses, and congestible services (non-excludable that can move from non-rival to rival) are in parallelograms. Bold boxes mark ecosystem services managed by a unique stakeholder group, and dashed boxes indicate ecosystem services used and impaired by the same single-stakeholder group. Numbers in parentheses indicate the stakeholder group (1=Primary sector; 2=Recreation sector; 3=Leisure; 4=Institutions; G=Global extent). The main clusters identified are marked in solid red boxes, and the secondary clusters in dotted red boxes.

stakeholders but contributed to the general degradation of the service. The impairers of habitat quality were leisure activities and the primary sector; however, they might not perceive it, as this service is not directly used but indirectly used through other services dependent on it.

Lastly, we identified a cluster that comprised those stakeholders having low use of ecosystem services and low ability to manage them. These were external to the ecosystem (e.g., the global food and raw materials markets and the carbon sequestration capacity of the atmosphere) (Fig. 6a, bottom left).

Key stakeholders control keystone ecosystem services: power matters

The identification of keystone ecosystem services to maintain the flow of services (Fig. 4), together with the identification of the stakeholder groups that used and managed them (Fig. 6) highlighted the critical effect of power asymmetries on access to ecosystem services. In our case study, the strongest power was held by the institutions group through the management of keystone supporting ecosystem properties and intermediate regulating services, on which many other services depend or that are used by most stakeholder groups. For instance, habitat quality had the strongest effects on the majority of ecosystem services: environmental education, recreation, and nutrient regulation (Fig. 4). Additionally, water-related services (water quality and freshwater supply) were the most conflicting services as they aggregated the largest number of beneficiaries and impairers (Fig. 6b). As a consequence, the institutions group had the power to promote synergies and trade-offs between ecosystem services and the power to limit the use of those services they managed to specific stakeholders (e.g., regulations on freshwater supply and fishing permits), excluding others, and thus, creating potential social imbalances.

Additionally, the recreation sector had strong power as they managed and used the cultural ecosystem services driving the economy of the area. Finally, the primary sector had intermediate but still important influence on some ecosystem services. For instance, they were able to moderately manage the provisioning services on which their main income is based (i.e., food production and raw materials). Interestingly, the SEM revealed that these services were fairly disconnected from other intermediate services (Table 3 and Fig. 4) because they were mainly dependent on human inputs (e.g., irrigation and fertilizers) rather than on ecosystem functioning. Additionally, some farming practices impaired critical ecosystem services (e.g., habitat quality, soil conditions, and nutrient regulation) that determined the integrity of intermediate and final services, thereby creating powerful feedback to the stakeholders using those intermediate and final services.

Discussion

Potential and limitations of the analysis of social-ecological interactions along the ecosystem services flow

Integrating both ecological and social interactions along the flow of ecosystem services is key to understanding the likely asymmetries between stakeholders fostered by environmental management and to promoting sustainable management of ecosystem services [69]. Recent research has increasingly been addressing the flow of ecosystem services from production (supply-side) to use by society (demand-side) [e.g., 5,36,70,71], and some have considered the implications of access to ecosystem services [72]. However, no framework has yet made explicit the existence of power relationships mediating both ecosystem services flows and stakeholder interactions. Specifically, recent research has pointed out the need to analyse the role of multiple stakeholder groups and their relationships with the provision, demand, and management of ecosystem services in order to contribute insights for sustainable management of ecosystem services [69]. In previous research, structural equation models have been used to test relationships between ecosystem properties and their effects on the provision of ecosystem services [73], and between ecosystem services and their effects on human well-being [74]. However, such analyses have not been previously connected to power relationships among stakeholders. Moreover, although Fisher et al. [75] discussed the importance of power related to poverty alleviation and ecosystem services, power relationships have still rarely been considered explicitly as a key factor determining asymmetries in the access to ecosystem services. Our study clearly demonstrates the relevance of power relationships in determining access to ecosystem services and its impact on the ecosystem services flow. Identifying and targeting such power relationships is essential for delineating environmental management policies while reducing trade-offs among ecosystem services [40] and thus, reducing social inequalities and conflicts.

The proposed framework was tested using our knowledge and data on the River Piedra case study, demonstrating its validity for uncovering the social-ecological interactions of ecosystem services. The application of this framework proved useful to identify: (i) keystone ecological properties and ecosystem services that determine the provision of other ecosystem services; (ii) relevant services for each stakeholder group; (iii) the ability of stakeholders to access and manage each service and their implication in the provision and use of other ecosystem services; and (iv) formal power asymmetries between stakeholders derived from their capacity for managing and using ecosystem services. Regarding the last two points, our analysis also detected which ecosystem services are managed by single-stakeholder groups, highlighting their strongest power in relation to other groups.

However, the application of such analyses in environmental management requires further work to specify and address the different types of power relationships between and within the stakeholder groups (e.g., property rights, family ties, prestige, age, gender, etc.). Other limitations of this study were: (1) it was time-consuming, as it required biological and social sampling; (2) difficulties connecting biological sampling with social sampling to give insights into power relationships; (3) difficulties including the adjacent municipalities within the geographic boundaries (i.e., the river basin) but outside the targeted ecosystem (i.e. the river floodplain) in the power relationships analyses.

The excludable and rival characteristics of ecosystem services

The classification of ecosystem services based on the concepts of excludability and rivalness (Table 4) concurred with most theoretical examples from Costanza [67] and Fisher et al. [3]. However, in our case study, we did not identify any examples of excludable but non-rival services, or non-excludable but rival, probably because management usually makes ecosystem services congestible (i.e., driving services from non-rival to rival). For instance, ecosystem services managed by the institutions group tended to fall into this category as this group can regulate the status of such services (e.g., policies to regulate water quality and waste water). Moreover, cultural services had two possible and opposite statuses: rival and excludable for activities performed on private sites or mediated by private companies (i.e., by the recreation sector), and non-rival and non-excludable, for those services enjoyed at open access sites. Hence, non-empowered stakeholders (i.e., the leisure group and the bulk of the primary sector) only had access to the remaining non-excludable and non-rival ecosystem services, and thus, are the most vulnerable stakeholders [68] at risk of being excluded from accessing the ecosystem services they need for their well-being.

These results contrast to other applications of this framework [76], indicating that the status of ecosystem services is highly context-dependent. In addition, classifying ecosystem services along the rival and excludable gradient enables accounting for the multiple possible statuses of an ecosystem service across land-use types and property rights. Indeed, the use of the land and access rights appeared as the critical factors determining the status of ecosystem services. Providing open access to lands and avoiding preventable dis-services (e.g., through conservation farming [77]) might change this classification. More importantly, focusing on managing the non-rival to rival movements of ecosystem services (i.e., the congestible services) would prevent their depletion.

Against monopolization of ecosystem services: insights for environmental management

As the proposed framework pinpoints, environmental management mediates the use of ecosystem services, and thus, their interactions [78]. Additionally, our results enabled us to distinguish how the governance for each ecosystem service may condition access/MAS BIEN SERÍA LA CALIDAD NO??? to it. For instance, single-stakeholder management systems in which ecosystem services are used and managed by a single stakeholder group generated positive feedback. Such positive feedback had two opposite effects: they either reinforced the service, for example, recreational activities managed by the recreation sector attracted more recreational activities; or depleted the service, such as for instance, soil conditions and nutrient regulation managed by the primary sector were consumed at faster rates than normal recovery; or the overuse of water by leisure activities impaired their initially good status. Although single-stakeholder management systems should ideally lead to negative feedback or internal self-regulation, there is a high risk of eliciting positive feedback, in which the service is depleted by unregulated use, decreasing the capacity of the system to supply services in the long term.

In addition, we identified top-down management systems, where management is made from the higher levels of governance – usually involving stakeholders external to the social-ecological system – to the local population. This was the case for the institutions group that managed habitat quality and recreation, and the recreation sector that managed cultural services. These management systems did not foster potential synergies among the ecosystem services supplied by the River Piedra, such as enhancing habitat quality and cultural services [28], and neither strengthened the communities' governance of their resources. Rather, the population in this area is mostly dependent on external capital such as the European Common Agricultural Policy subsidies for farmers, or the investments made by the main companies in the recreation sector. Top-down management systems often have low resilience [79] and can fail to resolve resource-users' conflicts [12]. However, examples of participatory bottom-up management systems such as decentralized forest management in Tanzania [80], coastal ecosystems in Kenya [81], and estuaries in South Africa [82] have proved to be important to complement existing top-down systems. In our case study and similar rural areas, such participatory systems could be implemented by local government and mediated by bridging institutions such as foundations and associations, adapting to the cultural and geographical characteristics of each social-ecological system. Encouraging participatory bottom-up management systems remain important because they reconnect the loop of the governance system from the top to the bottom and vice versa, enhance the ecological understanding of stakeholders, and foster more equal access to ecosystem services.

Conclusions

This paper shows that ecosystem services do not equally benefit the diversity of potential users, highlighting the importance of power relationships in ecosystem services' interactions and their influence on the flow of ecosystem services. The dependency relationships between ecosystem services stressed the importance of the use and management of keystone ecosystem services, i.e., those services that are essential for the provision of either intermediate or final ecosystem services. We identified the formal power relationships exerted by stakeholders according to their ability to access and manage ecosystem services, and the mechanisms they use to exert power. Therefore, those stakeholders able to manage such keystone ecological properties and ecosystem services can affect the well-being of other stakeholder groups by determining the ecosystem's capacity to provide services and/or by controlling access to them.

Consequently, in order to delineate sustainable management practices that foster equal access to ecosystem services, it is necessary to contribute detailed information on: (i) ecosystem services' interactions, (ii) ecosystem service beneficiaries, impairers, excluded, and managers, as well as (iii) the power relationships established among them. The present study presents a conceptual framework able to empirically operationalize the integration of such information.

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Supplementary material

S1. Methods used to sample ecosystem services.

Habitat quality

Riparian Quality Index: We used the Riparian Quality Index (RQI) (González del Tánago and García de Jalón, 2011) to assess habitat quality. The RQI evaluates seven riverbank attributes (i) dimensions of land with riparian vegetation (average width of riparian corridor); ii) longitudinal continuity, coverage, and distribution pattern of riparian corridor (woody vegetation); iii) composition and structure of riparian vegetation; iv) age diversity and natural regeneration of woody species; v) bank conditions; vi) floods and lateral connectivity; and vii) substratum and vertical connectivity), providing a relative score between 10 and 120 that was reclassified from 0 to 100. RQI was estimated in three plot replicates by land use type (i.e., dry cereal crops, irrigated cereal crops, poplar groves, fruit groves, riparian forests, abandoned crops, and urban areas) between July 2011 and July 2012.

Soil conditions

Soil formation: We measured the organic matter content in topsoil (0-10 cm) as an indicator of this service in July 2011 and July 2012. We sampled three patch replicates in each of the seven main land uses of our study area except in urban areas, where most of the soils are sealed (Felipe-Lucia et al., 2014). Three transects perpendicular to the river channel were established in each patch and three samples along each transect were taken at 1 m, 5 m, and 15 m away from the river. Half a kilogram of topsoil was collected at each point, dried (48 hours at 60ºC), sieved and milled. Total organic matter was analyzed using the LOI protocol (Lost On Ignition, Nelson and Sommers, 1996) and the average value (as soil weight percentage) of the two years was used as a measure of each sampling point.

Soil stability: We recorded the organic matter layer thickness in topsoil (0-10 cm) as an indicator of this service (Daily, 1997). We sampled three patch replicates by land use type except in urban areas, where most of the soils are sealed (Felipe-Lucia et al., 2014). Three transects perpendicular to the river channel were established in each patch and three measurements along each transect were taken at 1 m, 5 m, and 15 m away from the river. The organic matter layer depth (cm), excluding leaf litter, was recorded in the field with a measuring tape in July 2011 and July 2012, and average data of each point across both years were used as an indicator.

Water quality

Nitrite content in water (NO2-): We analyzed the nitrite content in water (ppm) as a measure of pollutant concentration in the river. 21 samples were collected monthly along the river in 2009. The sampling was designed to cover a wide range of situations representing the water quality of the study area and was repeated in specific months of 2010 and 2011 to account possible variation in the water flow rates. Samples were kept refrigerated and analyzed in laboratory within a week using standard methods (i.e., ionic chromatography (APHA, 1998)). Values per sample point were averaged across years and then by municipality. Because nitrite content in water is an indicator of pollutant concentration, we used the inverse values to account for water quality.

Nitrate content in water (NO3-): We analyzed the nitrate content in water (ppm) as a measure of pollutant concentration in the river. 21 samples were collected monthly along the river in 2009. The sampling was designed to cover a wide range of situations representing the water quality of the study area and was repeated in specific months of 2010 and 2011 to account possible variation in the water flow rates. Samples were kept refrigerated and analyzed in laboratory within a week using standard methods (i.e., ionic chromatography (APHA, 1998)). Values per sample point were averaged across years and then by municipality. Because nitrate content in water is an indicator of pollutant concentration, we used the inverse values to account for water quality.

Phosphate content in water (PO4-): We analyzed phosphate content in water (ppm) as a measure of pollutant concentration in the river. 21 samples were collected monthly along the river in 2009. The sampling was designed to cover a wide range of situations representing the water quality of the study area and was repeated in specific months of 2010 and 2011 to account possible variation in the water flow rates. Samples were kept refrigerated and analyzed in laboratory within a week using standard methods (i.e., ionic chromatography (APHA, 1998)). Values per sample point were averaged across years and then by municipality. Because phosphate content in water is an indicator of pollutant concentration, we used the inverse values to account for water quality.

Nutrient regulation

Total carbon in top soil (C): We measured the total carbon content in topsoil (0-10 cm) as an indicator of this service in July 2011 and July 2012. We sampled three patch replicates in each of the seven main land uses of our study area except in urban areas, where most of the soils are sealed (Felipe-Lucia et al., 2014). Three transects perpendicular to the river channel were established in each patch and three samples along each transect were taken at 1 m, 5 m, and 15 m away from the river. Half a kilogram of topsoil was collected at each point, dried (48 hours at 60ºC), sieved and milled. Total carbon was measured using a macro elemental analyzer (Vario Macro Max CN) and results were expressed in concentration (ppm). The average value (as soil weight percentage) of the two years was used for each sampling point.

Total nitrogen in top soil (N): We measured the total nitrogen content in topsoil (0-10 cm) as an indicator of this service in July 2011 and July 2012. We sampled three patch replicates in each of the seven main land uses of our study area except in urban areas, where most of the soils were sealed (Felipe-Lucia et al., 2014). Three transects perpendicular to the river channel were established in each patch and three samples along each transect were taken at 1 m, 5 m, and 15 m away from the river. Half a kilogram of topsoil was collected at each point, dried (48 hours at 60ºC), sieved and milled. Total Nitrogen was measured using a macro elemental analyzer (Vario Macro Max CN) and results were expressed in concentration (ppm). The average value (as soil weight percentage) of the two years was used for each sampling point.

Total phosphorus in top soil (P): We measured the soluble reactive phosphorus (SRP) content in topsoil (0-10 cm) as an indicator of this service in July 2011 and July 2012. We sampled three patch replicates in each of the seven main land uses of our study area except in urban areas, where most of the soils are sealed (Felipe-Lucia et al., 2014). Three transects perpendicular to the river channel were established in each patch and three samples along each transect were taken at 1 m, 5 m, and 15 m away from the river. Half a kilogram of topsoil was collected at each point, dried (48 hours at 60ºC), sieved and milled. SRP was extracted following the Olsen protocol (Olsen et al., 1954) and filtered. The extract was analyzed in an ionic chromatograph. The average value (as soil weight percentage) of the two years was used for each sampling point.

Biological control

Plant strata: To estimate the richness of plant strata we surveyed three plot replicates by land use type in July 2012. Urban areas were excluded as soils are sealed. Within each plot, three floodplain-wide transects (average transect length 57 m) perpendicular to the river channel were established 25 m apart. In each transect, we used the point-intercept method (Goodall, 1952) every 10 cm to estimate species occurrence and percent cover of each plant species (i.e., number of contacts relative to the total number of points sampled). Identification of plants at the genus or species level was corroborated using a regional herbarium (namely, herbarium of Jaca: http://proyectos.ipe.csic.es/herbario) and a botanist expert. Then, we classified vegetation records into four types of plant strata (namely, herb, creeper, shrub, and tree) and estimated the richness of plants strata using the vegan package (Oksanen et al., 2013) of the R software (R Development Core Team, 2013).

Carbon sequestration

Carbon sequestration: We used carbon (CO_2) sequestration of woody plants as a surrogate of this service (Trabucchi et al., 2014). Annual CO_2 sequestration rates by land use type were obtained from a national database (Montero et al., 2005; CITA, 2008) which estimated the amounts of carbon stored by above- and below-ground biomass of the main Spanish plant species and woody formations. Calculations are based on the species annual growth and transformed into CO_2 equivalent tons per hectare using stoichiometric equations (Montero et al., 2005). We used data from the closest plant species or woody formations to the land cover composition of our study area (e.g., average data of apple, pear, peach, and plum groves for fruit groves). Herbaceous species –and therefore, irrigated cereal crops and dry cereal crops – were not included because their annual CO_2 storage balance is null (CITA, 2008); for abandoned crops, only its woody formations (e.g., hawthorn) were considered. Urban areas have not been included either, since they act usually as a source of carbon rather than as a sink (but see Davies et al., 2011).

Freshwater supply

Water consumption: Water concessions of the River Piedra were obtained from public data provided by the water management body (Confederación Hidrográfica del Ebro, http://iber.chebro.es/webche/raInfo.aspx, accessed on 2011). Calculations were made to obtain the annual cubic meters supply at each municipality, which was used as an indicator.

Food production

Yield: We estimated the average yield (kilograms per hectare) of each of the main land use types of our study area from the latest update of a national public database (Instituto Nacional de Estadística, updated on 30.10.2012). We averaged irrigated wheat, barley, and corn yields to estimate food provision by irrigated crops; dry wheat, barley, and corn yields for dry cereal crops; and apple, pear, peach, and plum grove yields for fruit groves. The other land uses were assigned a yield value of 0.

Calories: Yield values for the crops growing within the study area were obtained from national databases statistics (Instituto Nacional de Estadística, updated on 30.10.2012), expressed as kilograms per hectare, and multiplied by the crop caloric value (kilocalories per 100 grams). The ecosystem service value is expressed as kilocalories per hectare (Felipe-Lucia et al., 2014).

Money: Crops productivity was calculated based on crops yield (Felipe-Lucia and Comín 2015) and the index of agricultural prices provided by the regional government (Gobierno de Aragón, http://www.aragon.es).

Raw materials

Production: We used the yearly aboveground dry biomass accumulation by land use type as a measure of the raw materials production. Values were obtained from a national database (Montero et al., 2005; CITA, 2008) that estimated the annual growth rates of woody species as tons of dry biomass per hectare, according to the average timber diameter. We adapted data from the closest woody species to the land cover composition of our study area (e.g., average data of apple, pear, peach, and plum groves for fruit groves). Herbaceous species –and therefore, irrigated cereal crops and dry cereal crops– were not included because their annual accumulated biomass balance is null (CITA, 2008), whereas for abandoned crops, only its woody formations (e.g., hawthorn) were considered. Note that biomass production is an indicator of the potential biomass provision by each land use type, thus referring to the potential use of the biomass as a raw material (i.e., making this use of the land incompatible with the provision of other services such as fruit production or carbon sequestration).

Aesthetics

Pictures: We counted the number of different people uploading pictures to Panoramio from each of the main land uses of each municipality within the floodplain of River Piedra. We used the finest resolution to count each single picture. This measure has been considered to be more appropriated than the total number of pictures, which would rather reflect the individual activity of photographers (Casalegno et al. 2013). Pictures focusing on buildings from all sorts (e.g., houses, towers, crosses, churches, hermitages, monasteries, etc.) were not considered because they were not directly related to any use of the ecosystem. The platform was accessed on 27.03.2014.

Recreation

Fishing: Available fishing stretches for recreational use at the River Piedra were obtained from the fishing regulatory policy of 2012 for the Autonomous Community of Aragon (BOA 2012) and drawn using GIS tools (ArcGIS 10.0, ESRI). Fishing available stretches were computed for both riversides. Stretches were converted into polylines, their perimeters calculated and summarized into stretches available or unavailable for fishing. Polylines were converted into polygons and intersected to the land use cover with a buffer of 10 m to add both the land use and municipality information of each stretch of the river. Then lengths were recalculated. The length of the river in each land use type of each municipality in relation to the total length of the river (i.e., including areas unavailable for recreational fishing) and in reference to a 1 hectare patch (a patch of 100 meters of side) was used as an indicator (i.e., Fishing at land use A = (Total length of land use A / Total length of the river) x 100) (Felipe-Lucia et al., 2014)).

Sports: Tracks of post-signed and user-designed paths were downloaded from both the regional tourist office website and wikilocs (http://senderos.turismodearagon.com and www.wikiloc.com, respectively; accessed on: 12.10.2012) following Trabucchi et al. (2014). Tracks within the study area were unified using GIS tools (QGIS, Quantum GIS Development Team), and overlapped to the study area viewshed. Then the viewshed of the shapefile obtained was calculated and intersected to the land use cover. Finally the extent of each land use that can be seen from the open-to-public used paths was calculated. Average values per hectare of each land use at each municipality were used as an indicator (Felipe-Lucia et al., 2014).

Picnic areas: We used the number of areas used for social amenity (e.g., picnic areas) within the study area as an indicator of this service (Posthumus et al., 2010). The number of areas in each municipality was counted by land use type in August 2012. To keep spatial scale consistency, these data were transformed into a density measure (i.e., Total number of picnic areas by land use type and municipality / Land use type cover extent at each municipality) (Felipe-Lucia and Comín 2015).

Environmental education

Educative panels: We used the number of educative panels with information about the ecosystem as an indicator of this service. This was the only available indicator distinguishing among land use types. Panels were counted in each municipality by land use type in August 2012. To keep spatial consistency, these data were transformed into a density measure (i.e., Total number of panels by land use type and municipality / Land use type cover extent at each municipality) (Felipe-Lucia and Comín 2015).

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S2. Stakeholders' interviews.

Table S2.1. Description of the respondents quoted in Table S2.2. Note that only a selection of the total sample is showed.

Code (Group_ intervieew)	Group	Function	Interview date	Interview length (h:mm)
1_1	Primary sector	Local farmer, retired	30.08.2011	1:07
1_2	Primary sector	Local shepherd	30.08.2011	1:31
1_3	Primary sector	Local farmer	06.08.2011	1:22
1_4	Primary sector	Local farmer	23.03.2012	1:01
1_5	Primary sector	Local farmer	31.08.2011	1:11
1_6	Primary sector	Local farmer	31.08.2011	1:34
1_7	Primary sector	Local shepherd, retired	03.08.2011	0:30
2_1	Recreation sector	Local lodge owner	30.08.2011	0:35
2_2	Recreation sector	Local nature business	01.08.2011	0:58
2_3	Recreation sector	Camping owner	26.08.2011	1:05
2_4	Recreation sector	Local hotel owner	04.08.2011	0:45
2_5	Recreation sector	Adventure enterprise partner	2.10.2011	0:39
2_6	Recreation sector	Local lodge owner	30.08.2011	1:47
3_1	Leisure	Local seasonal resident (weekends, holidays, etc.)	03.08.2011	2:40
3_2	Leisure	Local seasonal resident (weekends, holidays, etc.)	07.03.2012	1:14
3_3	Leisure	Local seasonal resident (weekends, holidays, etc.)	01.08.2011	1:04
3_4	Leisure	Seasonal resident (weekends, holidays, etc.), retired	02.08.2011	0:54
3_5	Leisure	Local seasonal resident (weekends, holidays, etc.)	28.09.2011	0:31
3_6	Leisure	Permanent resident, retired	5.10.2011	1:15
4_1	Institutions	Environmental technician (engineer) working on several environmental projects on the area (e.g., bioengineering)	20.02.2012	0:50
4_2	Institutions	Local council, councillor	05.08.2011	1:00
4_3	Institutions	Local council, mayor	05.08.2011	1:00
4_4	Institutions	Elementary school teacher	05.10.2011	1:14
4_5	Institutions	High school teacher	05.10.2011	0:59
4_6	Institutions	University professor	22.02.2012	1:23
4_7	Institutions	Local council, mayor	05.08.2011	0:30

Table S2.2. Ecosystem services co-produced, used, and impaired by each stakeholders group. Citations are in italics, and numbers in parentheses indicate the code of the interviewee, described in Table S2.1. Acronyms: CHE=Confederación Hidrográfica del Ebro (Regional water management body of the Ebro basin); MMA=Ministerio de Medio Ambiente (Ministry of the Environment).

Ecosystem service	**Stakeholder group**	**Co-produced by**	**Used by**	**Impaired by**
Soil conditions	1- Primary sector		Famers use soil properties to growth their crops. For instance, they benefit from organic matter content and layer's thickness. *"Soil is the most important, when it is flooded gives 25% more yield the next year. It is good for farming because the sediments are good*" (3_1)	Farmers' practices (e.g., tillage) erode soils by oxidizing the organic matter and breaking soils' structure. "*Now they [the farmers] use herbicides. They say everything is cleaner, but it breaks riverbanks and the land crumbles*" (1_2)
Habitat quality	1- Primary sector		[Used indirectly]	Usually farmers prefer clear riverbanks to avoid shading diminishing crops yield. In consequence, they used to cut trees and avoid new planting. "*We used to clean the riverbanks, we cut tree branches and the grass*" (1_3) *"We do not want big trees because they do not allow crops to grow up, and we need yields to make money. It's our job, and some years are good and some others are worse"* (1_5) "*If we clean often we also destroy the river ecological system, because there used to be pools but now they are clogged*" (4_2)
	2- Recreation sector		[Used indirectly]	
	3-Leisure		[Used indirectly]	Some fishermen leave waste behind close to fishing areas, and break trees' branches

Ecosystem service	Stakeholder group	Co-produced by	Used by	Impaired by
				to access the river. "*People are not aware of the value we have here. There is a lack of awareness and environmental education*" (3_2)
	4-Institutions	Scientists, technicians and the government contribute to enhancing this service by ecological restoration projects. "... *yes, protection of the aquifer, and the landscape is possible and it has to come from the CHE and the MMA*" (3_2) "*The project was ordered by the MMA and the CHE, so the most important issues here were water, soils, plants, and animals. Productive uses are not so important, neither tourism. Also the landscape, educational issues, social relationships, and life quality. That would be the scheme for the MMA, which I followed*" (4_1)	[Used indirectly]	
Water quality	1- Primary sector		The fish farm benefits from the quality of the water sources of the River Piedra, which makes trout to grow up better than in other rivers.	Farmers impair water quality because of diffuse pollution and crops' run-off. The fish farm pollutes the water by direct discharge.

Ecosystem service	Stakeholder group	Co-produced by	Used by	Impaired by
			"*The water of our river is so good that the fish farm grows here the juvenile fishes. Each week, a lorry collect 25000 kg of trout and go to another place to grow up to fatten*" (3_3)	*"Water has such an anise color. It was clear before, but now it is waste water from the fish farm and Cimballas' houses, and some other chemical fertilizers (…) Last year I drank water from the river and I had to stay in bed for three days, with gastroenteritis, now you cannot drink water from any place"* (3_1) *"There were not crops, nitrates, sulphates, etc. We have destroyed the river, because the easiest thing it is to use chemical herbicides....but if the Ministry of Agriculture allows us to use it...and also the fish farm of Cimballa, it had not wastewater treatment plant and discharged some wastewater into the river"* (1_6)
	2- Recreation sector		They benefit from performing activities in a clean water river, which attract tourists to come. "*The economy of this area depends upon the river, the Monasterio de Piedra, the spa resorts, and also most of the people depend on them*" (4_4) "*It's trendy to attract rafting and kayaking tourists*" (1_2) "*The main attraction is the Monasterio de Piedra, but there are also the sources of the River Piedra (…) and the reservoir (…)*" (2_1)	Hotels impair water quality by the discharge of untreated wastewater. Their daily activities pollute water because of the lack of functional wastewater treatment plants in the villages, which spill out directly to the river. This fact worsens during summer months, when population doubles and the water flow decreases, giving as a result a bad colored and smelly river. *"In LLumes there is any waste water collection, so everything goes directly into the river and it can be dangerous because*

Ecosystem service	Stakeholder group	Co-produced by	Used by	Impaired by
				we are also drinking from wells" (1_4) *"Because the waste water from Carenas spills directly to the river and in Castejón, too… and then if the river water flow is low it smells badly"* (4_7)
	3-Leisure		Tourists benefit from having a clean water river to enjoy leisure activities (e.g., relaxing, kayaking, and fishing). "*This stretch of the river is really good, I do not see dirty water, I see very clear water*" (4_2)	
	4-Institutions	The central government is in charge of providing wastewater treatment plants to villages, and local governments are in charge of assuring their proper functioning. The CHE should care about its quality. *"The wastewater collection has been demanded several years ago, but nobody does a thing. We are systematically mistreating the river*" (1_4)	[Used indirectly]	
Nutrient regulation	1- Primary sector		Famers benefit from natural nutrient regulation to growth their crops. For instance, they benefit from organic carbon, nitrogen, and phosphorus content.	Farmers' practices (e.g., tillage) erode soils by oxidizing the organic nutrients. Additionally, the use of chemical nutrients deregulates natural nutrient cycling. *"Now they [the farmers] use chemical*

Ecosystem service	Stakeholder group	Co-produced by	Used by	Impaired by
			"*Soil is the most important, when it is flooded gives 25% more yield the next year. It is good for farming because the sediments are good*" (3_1)	*fertilizers and before they used manure"* (3_3)
Biological control	1- Primary sector		Farmers benefit from natural biological controllers such as birds and other insects. *"There were understory formations that hosted many different bird species and also snakes and other animals"* (1_2)	Farmers impair this service because of the use of chemical herbicides and other pesticides which are not specifically targeted for undesirable species for farming, thus, affecting the natural regulation of the ecosystem. *"Now they [the farmers] use herbicides. They say everything is cleaner, but it breaks riverbanks and the land crumbles*" (1_2). "*There were not crops, nitrates, sulphates, etc. We have destroyed the river, because the easiest thing it is to use chemical herbicides....but if the Ministry of Agriculture allows us to use it....*" (1_6)
	3-Leisure		People benefit from natural biological controllers such as birds and insects eating mosquitos and crops' pests. "*Here there are not many mosquitos because the temperature of the water controls that, and here the water is fresh*"(3_4)	
Gas regulation	1- Primary sector	Fruit groves and poplar groves owners could contribute to this service by the amounts of carbon		Farmers' practices (e.g., tillage, use of chemical pesticides, etc.) can liberate carbon to the atmosphere. However, the potential harm caused by farmers is just a

Ecosystem service	Stakeholder group	Co-produced by	Used by	Impaired by
		sequestered by their trees. *"We are planting our lot with walnut trees (…) others have poplars"* (3_2)		little contribution to the global problem, which means farmers do not directly affect other users but rather they contribute to the general degradation of this service. [Not perceived by the interviewees]
	2- Recreation sector	Tree plantations made by the Monasterio de Piedra a hundred years ago contribute to carbon sequestration. [Unrecorded communication]		
	4-Institutions	Ecological restoration projects comprising tree plantations (financed by the MMA and assessed by scientists and technicians) contribute to carbon sequestration. *"They [the MMA] should do more plantations to have more trees"* (1_7)	[Used indirectly]	
Food provision	1- Primary sector	They produce food from crops and trout farming. "*There are self-consumption, and some professional farmers*" (2_2)		
Raw materials	1- Primary sector	Poplar groves owners produce wood.		

Ecosystem service	Stakeholder group	Co-produced by	Used by	Impaired by
		"Poplars used to be cut down and the wood was used to build houses" (3_1)		
Freshwater supply	1- Primary sector		Farmers benefit from access to water from the river and ditches. The fish farm takes freshwater directly from a source. "*The waterwheels allow to divert water to the ditches"* (2_1) "*I have irrigated lands and turn dry lands into irrigated with the water from the river"* (1_4) "*All the water used by the fish farm comes from the sources*" (2_3)	The fish farm has exclusive use of a water source, impeding other users to access to it. Water demand by farmers out of the catchment decreases the amount of water remaining in the river for other uses. "*There is another source close to the road (…) there was a waterfall (…) but the fish farm built a wall and now it is stored (…). You can still see the water, but it is not so beautiful*" (1_5) "*When the reservoir was built, everybody had to emigrate (…), and now the farmers from La Almunia, are benefiting from that richness, (…) and control the reservoir and we can do nothing*" (3_3)
	2-Recreation sector		Hotels use the water for domestic use. Adventure companies benefit from high flows to perform their activities in the river. The Monasterio de Piedra also benefits from higher flows increasing the aesthetic impact of the waterfalls. *"With the idea of having higher water flow and less water loss... the river is channelized and then there is more water in the Monasterio de Piedra waterfalls (…) 90% of the River Piedra is currently used to supply water to the*	

Ecosystem service	Stakeholder group	Co-produced by	Used by	Impaired by
			waterfalls of the Monasterio de Piedra. They are nothing without water and with water they are a company making a lot of money. They are a private company using a public good -the water- to make money” (1_4)	
	3-Leisure		The high flow levels benefit clients of adventure companies. *“The River Piedra has a large and continuous flow; even when there are droughts here there is always water”* (1_3)	
	4-Institutions	The CHE regulates the use of the water. *"… and then they [the CHE] cut down the river, by the end of September they close the dam and the river flows with a very little water*” (1_7) *“They [the CHE] cut down the release from the reservoir and the water flow in the river is left at a minimum level, which is too low”* (4_7)	[Used indirectly]	The CHE entitles to the use of water, which can create disagreements or inequalities among users. *“In Castejón and Carenas we have a concession (from the CHE) of free water for irrigation”* (1_7) *“We could irrigate more lands but the CHE does not give more concessions”* (1_4) *"The irrigation function of the river, how is it? it is how downstream people wants: the water retained in the reservoir*" (1_2)
Aesthetics	1- Primary sector		They enjoy the beauty of the place where they live. "*The water source of the "eyes" is wonderful. And it is so deep, more than*	

Ecosystem service	Stakeholder group	Co-produced by	Used by	Impaired by
			5 meters. It is so wonderful to see how the water comes up gushing" (1_5).	
	2- Recreation sector	The Monasterio de Piedra contributes to this service by the maintenance and enhancement of the waterfalls and tree cover. *"They [the Monasterio de Piedra] diverted the water flow to have more water in the waterfalls"* (1_4)	The aesthetics value of the area is the main attraction for tourists. "*People likes it, it is a natural park very nice, people loves it... you come here and you see a natural park with waterfalls..."* (2_4)	
	3-Leisure		They enjoy the beauty of the place where they perform their leisure activities "*The River Piedra has many charming beauty spots*" (3_1) *"Look at the picture with the watermill, you'll see what a wonderful landscape makes the river... look this riverine landscape.... we have many pictures of the river and the bridge in the brochures of the village festival"* (4_3)	
Recreation	1- Primary sector		Local people use the area for personal recreation. "*The fact that you can go for a walk along the river every morning or every afternoon is pure enjoyment, and it is only appreciated by those loving nature.*" (3_3)	
	2- Recreation	They offer recreational	They benefit from having clients visiting	

Ecosystem service	Stakeholder group	Co-produced by	Used by	Impaired by
	sector	activities *"The project is about small rafting boats and kayaks, maybe hydrospeed, (…) and also climbing in Nuévalos"* (2_5)	the area for recreational activities. "*People likes it, it is a natural park very nice, people loves it... you come here and you see a natural park with waterfalls...*" (2_4) *"I walk many, many, many days in the afternoon by the riverside with my dogs. It's a very pleasant walk, fresh without sun beats, and very good"* (2_4)	
	3-Leisure		The main reason of visiting the area is for performing recreational activities (e.g., sports, fishing, picnicking). *"People like going for a walk along the river, people likes it, and then we have a reservoir, a picnic area, we have a very beautiful landscapes here, I like them. Yes, it is important because it is a beautiful landscape and people comes and watch it but it gives any money to the village; it is more a moral benefit."* (4_3) *"There is a place for bird watching, if you like, you can enjoy the landscape, the architecture, oenology, customs, etc."* (3_6)	
	4-Institutions		[Used indirectly]	Some fishermen agree that several actions performed by the CHE have caused a decrease in trout. *"The fisheries were destroyed when they [the CHE] cleaned the river, they destroyed*

Ecosystem service	Stakeholder group	Co-produced by	Used by	Impaired by
				all the fisheries and now there are some trout, but only a few" (1_1) *"There is no more sand where it used to be; now there is only that black mud… and trout is very delicate and if it is not good she doesn't raise juveniles...*" (2_4)
Environmental education	2- Recreation sector	Nature/adventure companies usually are pro-environmental education, so producing the service. The Monasterio de Piedra has many panels providing information about the functioning of the ecosystem. *"They [the companies] explain the uses of the water in the River Piedra, the trees, the birds, and many other things"* (3_6)	Nature/adventure companies usually are pro-environmental education and use the educative panels and other facilities. *"Now there is a sighting hut and some panels with the birds we can observe from here"* (2_2)	
	3-Leisure		People can learn from the educative panels. *"Now there is a sighting hut and some panels with the birds we can observe from here"* (2_2)	
	4-Institutions	Local councils and the MMA (advised by scientists) have provided the area with panels that explain the ecosystem functioning of the	Scholars and scientific groups visit the area to learn about the ecosystem and benefit from the existing panels. "*We teach about trees, learning their names, making the difference among*	

Ecosystem service	Stakeholder group	Co-produced by	Used by	Impaired by
		area *“We have created a rest area close to the road and from there you can see a wonderful landscape of the valley"* (4_3)	*them and their fruits.... we also do orienteering, trekking, and even clean the river and learn to respect the river*" (4_4) "*In the Piedra valley what is important is geology. There are huge ripples formations, fossils around Carenas and Castejón. And then in the gorges you have old hives, etc."* (4_5) *"Torralba has been more promoted and people from the university has come to visit the gorges and surroundings"* (3_5) *“We have many publications about the River Piedra. We have investigated fossil and current tufa since the last 12 years. And we teach about it at the University. We have also organized a conference where we visited the area”* (4_6)	

S3. Classification of ecosystem services along the rival/excludable gradient

We observed that provisioning services (food production, raw materials, and freshwater) were rival and excludable services, because they were provided in particular (private) sites or given in concession to private operators. Food production and raw materials were excludable because they were obtained on private land and rival because their consumption made them less available for others. Freshwater supply was partially considered a non-rival and non-excludable service, but the fact that the amounts of water in a river are limited, made it a rival service. Moreover, as water intake is regulated by the regional water management body (a government office), its use can be allowed or restricted to certain petitioners by water concessions, becoming an excludable service.

Supporting ecosystem properties and most regulating services were non-excludable, as they have public access. However, these can be congestible given the limited capacity of ecosystems to buffer their overuse; for instance, streams have a limited capacity for maintaining good water quality after their use. Soil conditions, nutrient regulation, and habitat quality were partially included in this category, because by providing rights to access the land, these services are free to use (non-excludable); however, land property rights can exclude some stakeholders from them. In addition, high levels of use can deplete soil good conditions and the capacity to regulate nutrient cycling, causing a decrease in habitat quality (i.e., becoming a congestible service). Exceptions were biological control, which was considered non-rival because it cannot be consumed, and carbon sequestration, which at the small spatial scale of this study is not congestible.

Cultural ecosystem services (aesthetics, recreation, and environmental education) were partially non-rival (i.e., can benefit many people at the same time without being consumed) and non-excludable, as enjoying the landscape is usually open access. However, these can also be excludable, as some specific sites in our study area required an entry ticket or permit (e.g., fishing permits, entry to Monasterio de Piedra), and rival when available in a limited space or if an excessive number of users decrease their value.

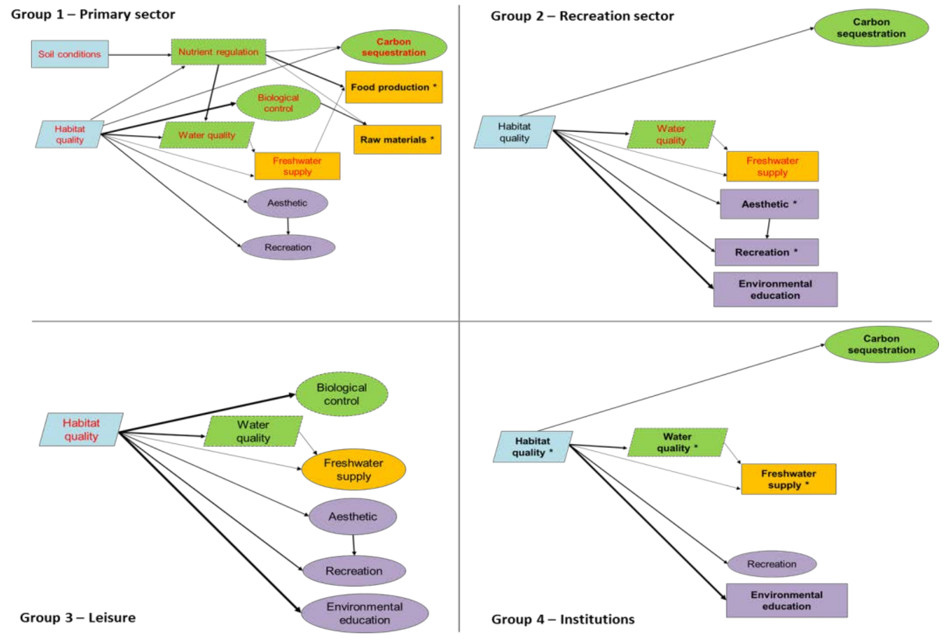


Figure S3. Ecosystem services classified as excludable/rival and related to each stakeholder group. Colors indicate the type of ecosystem services (green=regulating, gold=provisioning, purple=cultural) and supporting ecological properties (blue). Intermediate regulating services are dashed and final services are solid. Rival and excludable services are in rectangles, non-rival and non-excludable services are in ellipses, and congestible services (non-excludable that can move from non-rival to rival) are in parallelograms. Impaired ecosystem services are in red, ecosystem services managed or co-produced are in bold, and they are marked with an asterisk (*) when managed by a single group. Note that habitat quality and carbon sequestration were only indirectly used by groups 1, 2 and 3, and that all ecosystem services linked to group 4 (excluding environmental education) were used indirectly.

CAPÍTULO 7. APORTACIONES PARA LA GESTIÓN DEL VALLE DEL RÍO PIEDRA A PARTIR DE MODELOS ALTERNATIVOS DEL USO DEL SUELO*

RESUMEN. Este capítulo aporta un ejemplo de la valoración de los servicios de los ecosistemas aplicado a la gestión del valle del río Piedra. Consiste en un análisis preliminar de escenarios de gestión alternativos como base para la elaboración de escenarios de futuro de manera participativa. Se plantean cinco posibles escenarios de futuro resultado de simular un gradiente de intensificación del uso del suelo (desde el abandono rural a la intensificación de la agricultura), y un gradiente de restauración del bosque de ribera (de la situación actual a la restauración ecológica del bosque de ribera). En cada uno de los cinco escenarios se han valorado 29 indicadores de servicios de los ecosistemas incluyendo servicios de regulación, abastecimiento, culturales y de soporte. Para identificar asociaciones de servicios de los ecosistemas en relación a los escenarios de futuro, se realizó un análisis de redundancia (RDA). Los resultados mostraron de manera consistente una clara oposición entre municipios cuya llanura de inundación conserva el bosque de ribera frente a los que está dedicada fundamentalmente a la agricultura, y dentro de ésta, la dedicada al secano frente al regadío. EL RDA también permitió identificar dos asociaciones de servicios de los ecosistemas relacionados con actividades concretas: una relacionada con la producción de bienes materiales y otra con las actividades socio-culturales. Los resultados muestran que, en el valle del río Piedra, el escenario de *Conservación & Producción*, proporciona un conjunto de servicios de manera equilibrada, ya que combina la conservación y mejora de los servicios culturales, de soporte y de regulación asociados al escenario *Conservación & Restauración*, y los beneficios derivados de la producción agrícola asociados al escenario *Agricultura intensiva*. La comparación entre estos cinco escenarios de futuro (situación actual frente a cuatro escenarios alternativos) ha permitido mostrar el efecto de distintas alternativas de gestión en la provisión de servicios de los ecosistemas a escala de paisaje y la necesidad de actuar en esta zona para evitar la pérdida generalizada de servicios.

Palabras clave: escenarios de futuro, servicios de los ecosistemas, llanura de inundación, paisaje multifuncional, gestión de ecosistemas.

* Modelling future scenarios for ecosystem services management (Felipe-Lucia and Comín, *in prep.*)

Introducción

Este capítulo pretende aportar un ejemplo de la valoración de los servicios de los ecosistemas aplicada a la gestión del valle del río Piedra. Para ello, se plantean cinco posibles escenarios de futuro en función de la intensidad del uso del suelo de la llanura de inundación (Tabla 1). Los escenarios de futuro son una herramienta muy utilizada en la gestión del territorio para identificar ventajas e inconvenientes de las diferentes alternativas de gestión. En este proceso, que implica tanto la definición de alternativas como la valoración de las mismas, suelen participar los agentes sociales de interés del territorio (Vermaat *et al.* 2012; Liekens *et al.* 2013; Labiosa *et al.* 2013; Lamarque *et al.* 2013; Priess and Hauck 2014). Existen distintas metodologías para la elaboración de escenarios, siendo cada vez más común la valoración de los servicios de los ecosistemas (García-Llorente *et al.* 2012; Plieninger *et al.* 2013; Reed *et al.* 2013), pues permiten captar un espectro más amplio de los efectos de los escenarios que los basados únicamente en aspectos económicos o sociales. En este capítulo se realiza un análisis preliminar de escenarios de gestión alternativos que pueda servir de base para la elaboración de escenarios de futuro de manera participativa.

Métodos

La definición de los escenarios se ha basado en el conocimiento adquirido de la zona de estudio, así como en una revisión de la literatura referente a la elaboración de escenarios de futuro. En este caso, los escenarios son el resultado de simular un gradiente de intensificación del uso del suelo (desde el abandono rural a la intensificación de la agricultura), y un gradiente de restauración del bosque de ribera (de la situación actual a la restauración ecológica del bosque de ribera). En cada uno de los cinco escenarios se han valorado 29 indicadores de servicios de los ecosistemas (Tabla 2), de los cuales 14 corresponden a servicios de regulación (regulación del ciclo de nutrientes, del clima, de la calidad del agua, control biológico y secuestro de carbono), 6 a servicios de soporte (formación del suelo y provisión de hábitat), 4 a servicios de abastecimiento (producción de alimentos y materias primas) y 5 a servicios culturales (uso recreativo, educación ambiental y disfrute estético). La selección de los indicadores utilizados se ha basado en los siguientes criterios (van Oudenhoven *et al*. 2012):

- Espacialmente definidos, escalables y cuantificables
- Sensibles a los cambios del uso del suelo
- Relación clara entre el indicador y el servicio del ecosistema
- Fácilmente comprensibles por expertos y no-expertos
- Disponibilidad de datos

Tabla 1. Argumento de los escenarios y porcentaje de suelo ocupado por el bosque de ribera, suelo agrícola y cultivos abandonados.

Argumento	Bosque ribera	Suelo agrícola	Cultivos abandonados
Escenario 0 - *Continúa la situación actual* La llanura de inundación del río Piedra se caracteriza por un uso agrícola (60%), compuesto de cereal de secano en la parte alta, cereal de regadío y cultivo de choperas en la parte media, y frutales y huertas en la parte baja. Una parte importante de las zonas de cultivo están sin cultivar (16%) y el embalse de la Tranquera, construido en 1959 entre los municipios de Nuévalos, Carenas e Ibdes, cubre las tierras antaño más fértiles. El bosque de ribera natural se restringe a las Hoces del río Piedra, situadas entre las poblaciones de Aldehuela de Liestos y Embid. Además, el parque natural privado del Monasterio de Piedra, en el municipio de Nuévalos, constituye un importante espacio verde tanto por el contraste con el paisaje semiárido que lo rodea como para la economía local. El turismo generado por este parque constituye el principal motor económico de la zona que, centrado en los meses de verano y fines de semana, atrae a los turistas a otras actividades relacionadas con la naturaleza en los alrededores, dando uso a hoteles, restaurantes, casas rurales y campings. También existe una pequeña central hidroeléctrica propiedad del Monasterio de Piedra y una piscifactoría en Cimballa. El pastoreo en esta zona está muy reducido (uno o dos pastores con muy pocas cabezas de ganado por municipio). Las empresas de aventura tienen un importante potencial pero todavía no se han desarrollado.	1,6 %	43,6 %	15,9 %
Escenario 1 - *Conservación & Restauración ecológica del bosque de ribera* Se conservan o restauran los 5 metros de dominio público hidráulico (DPH, Real Decreto 9/2008) en ambas márgenes del río Piedra, así como los espacios incluidos en la categoría LIC (Lugar de Importancia Comunitaria): Hoces de Torralba – río Piedra, Lagunas y parameras del señorío de Molina y Riberas del Jalón (Bubierca – Ateca). Además, se convierten en bosque de ribera los campos abandonados que superan las 0,5 hectáreas de superficie. En estos espacios se plantan especies típicas de los bosques de ribera (*Salix* sp., *Populus* sp., *Fraxinus* sp., etc.), logrando una reducción del 90% de los contaminantes (nitratos, nitritos, fosfatos, sulfatos, materia orgánica y sólidos en suspensión) que llegan al río por escorrentía superficial y sub-superficial (Parkyn 2004). En las zonas restauradas se permite el acceso público, desarrollándose el turismo de naturaleza centrado en la educación ambiental e incrementándose las visitas de grupos senderistas, la ornitología y la pesca. Las zonas cultivadas mantienen su misma actividad pero se eliminan los azudes no funcionales.	18,2 %	41,1 %	2,8 %

Argumento	**Bosque ribera**	**Suelo agrícola**	**Cultivos abandonados**
<u>Escenario 2 - *Intensificación de la agricultura*</u> Se incrementa la producción agrícola poniendo en cultivo todos los campos abandonados, destinándolos a cereal de secano en la cuenca alta (aguas arriba de Aldehuela de Liestos). En la cuenca media y baja (aguas abajo de Cimballa), se transforman a regadío tanto los secanos como los campos abandonados, dedicando un tercio de cada municipio a cereal de regadío, un tercio a frutal y otro tercio al cultivo de choperas, en función de las facilidades de cada propietario para llevar a cabo un tipo u otro de transformación. El uso de fertilizantes y plaguicidas de origen químico sintético asociado a los cultivos de regadío provoca un incremento en la concentración de estos contaminantes (nitrato, nitrito, fosfato, sulfato) en las aguas del río estimado en un 20% (Darwiche-Criado *et al.* [*en revisión*]). Al aumentar la presión sobre el río, disminuye el incipiente turismo de naturaleza y la pesca queda relegada al embalse de la Tranquera.	1,6 %	59,4 %	0,0 %
<u>Escenario 3 - *Conservación del bosque de ribera & Producción agrícola*</u> Se conservan o restauran los 5 metros de DPH (Real Decreto 9/2008) en ambas márgenes del río Piedra y los espacios incluidos en la categoría LIC. En estos espacios se plantan especies típicas de los bosques de ribera (*Salix sp., Populus sp., Fraxinus sp.,* etc.) y se permite el acceso público. De esta manera se alcanza una reducción del 90% de los contaminantes que llegan al río por escorrentía superficial y sub-superficial (Parkyn 2004). Los campos abandonados se ponen en cultivo, destinándolos a cereal de secano en la cuenca alta (aguas arriba de Aldehuela de Liestos). En cada municipio de la cuenca media y baja (aguas abajo de Cimballa), se transforma un tercio de los campos abandonados a cereal de secano, un tercio a frutal y otro tercio al cultivo de choperas. Las zonas cultivadas previamente mantienen su misma actividad pero se eliminan los azudes no funcionales. El turismo de naturaleza centrado en la educación ambiental se desarrolla, y se incrementan las visitas de grupos senderistas, la ornitología y la pesca. Se fomenta la implantación de empresas de aventura (escalada, rafting, kayak). Se recuperan las infraestructuras hidráulicas tradicionales (norias, molinos) y las visitas de etnoturismo complementan la oferta de actividades en la zona. Prolifera la recuperación de casas para convertirlas en casas rurales.	9,5 %	52,6 %	0,0 %
<u>Escenario 4 - *Abandono rural*</u> Se mantiene la situación actual de los bosques de ribera y se abandonan los cultivos menos productivos, por lo que en la cuenca media y baja (aguas abajo de Cimballa) sólo permanecen los actuales cultivos de cereal de regadío y choperas de plantación y desaparecen el resto de cultivos (frutales, cereal de secano, etc.). En la cuenca alta (aguas arriba de Aldehuela de Liestos) sólo permanece el cereal de secano y desaparecen el resto de cultivos (frutales, choperas de plantación, etc.). Es decir, se mantiene la inversión actual en agricultura pero no se invierte en nuevas infraestructuras ni maquinaria. La reducción de la presión agrícola reduce un 40% la carga de contaminantes en el agua del río.	1,6 %	39,2 %	20,2 %

Tabla 2. Indicadores utilizados

Categoría	Servicio	Indicador	Unidades
Soporte	Estabilidad del suelo	Espesor capa de materia orgánica	m^3
	Calidad del hábitat	Riparian Quality Index	-
Regulación	Calidad del agua (depuración del agua)	Materia orgánica disuelta	ppm
		Amonio disuelto	ppm
		Nitrito disuelto	ppm
		Nitrato disuelto	ppm
		Sulfato disuelto	ppm
		Fosfato disuelto	ppm
		Sólidos en suspensión	ppm
	Formación de suelo	Contenido en materia orgánica	kg
	Regulación de nutrientes	Contenido en carbono	kg
		Contenido en nitrógeno	kg
		Contenido en fósforo	kg
	Regulación del clima	Variación de la temperatura	°C
		Variación de la humedad	%
	Control biológico de plagas	Estratos de vegetación	nº
	Secuestro de carbono	Secuestro de carbono	CO_2eq
Abastecimiento	Producción de alimentos	Euros	€
		Calorías	kcal
		Productividad	kg
	Producción de materias primas	Acumulación de biomasa	T
Cultural	Valor estético	Densidad de fotos	nº/ha
	Recreativo	Densidad de sitios	nº/ha
	Educación ambiental	Densidad de paneles educativos	nº/ha
	Deportes	Densidad de rutas	m/ha
	Disfrute de la naturaleza	Superficie forestal	m^2

Para obtener un valor total de los servicios de los ecosistemas en función de la superficie ocupada por cada uso del suelo en cada escenario, es decir, a escala de paisaje, se siguieron métodos diferentes según el origen de los datos. En el caso de indicadores cuantificados por uso de suelo, se estimó su valor medio por unidad de superficie y se multiplicó por la extensión de cada uso del suelo en cada municipio. En el caso de indicadores medidos a escala municipal, se agregaron los valores por municipio, y para los datos a escala de paisaje, se tomó ese valor directamente (ver detalles de los métodos en los apéndices de los capítulos 3 y 6). En este estudio solamente se han tenido en cuenta los valores derivados de los principales usos del suelo de la llanura de inundación, excluyendo usos del suelo secundarios como frutal de secano y forestal, cuya contribución es mínima, aunque incrementarían ligeramente los valores totales a escala de paisaje. Los datos se han ajustado a situaciones realistas de gestión; por ello, la producción de materias primas se ha considerado únicamente como la producida en las choperas de plantación, y no la madera que podría extraerse de los frutales y bosques de ribera, ya que esos usos del suelo dejarían de valorarse para el resto de servicios de los ecosistemas.

Para identificar posibles sinergias y antagonismos entre servicios de los ecosistemas derivados de cada uno de los escenarios propuestos, se utilizó el modelo de ecuaciones estructurales desarrollado en el capítulo 6 (Felipe-Lucia et al. [*en revisión*]). En este caso, únicamente contamos con referencias que avalan el incremento o reducción de la calidad del agua derivada del cambio de uso del suelo. Por tanto, solamente se tuvieron en cuenta los efectos sinérgicos de la recuperación de riberas en la mejora de la calidad del agua por reducción de la carga de nitratos, y la pérdida de calidad del agua derivada de la intensificación agrícola (Tabla 1). La representación de los escenarios, así como los cálculos de superficies para cada uso del suelo se efectuaron con ArcGIS 10.2 (ESRI).

Los indicadores de servicios de los ecosistemas más representativos fueron seleccionados en base al coeficiente de correlación de Pearson (r) calculado mediante el software R (R Core Team). Para un mismo servicio, los indicadores altamente correlacionados ($r>0.7$) se descartaron y el resto se representaron en un diagrama radial, en el que los ejes representan los servicio de los ecosistemas y las líneas el valor que alcanzan en cada escenario (Tallis *et al.* 2008; Butler *et al.* 2013). Para calcular el valor de cada servicio de los ecosistemas en cada escenario, se sumaron los valores por uso del suelo para cada servicio y se normalizaron los valores de manera que el valor máximo fuera 1 y el mínimo 0. En el caso de los servicios de regulación del clima y calidad del agua, se calculó el inverso para que valores más altos indicaran mayor provisión de servicio. Finalmente, para identificar asociaciones entre servicios de los ecosistemas en relación a los escenarios de futuro, se realizó un análisis de redundancia (RDA), que combina los análisis de regresión con el análisis de componentes principales (ACP), utilizando el paquete ‘vegan’ de R (Oksanen et al. 2013). Para ello, se utilizaron los servicios de los ecosistemas no correlacionados, agregando los valores de cada servicio de los ecosistemas por municipio.

Resultados

La figura 1 muestra un ejemplo del efecto de cada uno de los escenarios de gestión alternativos en la composición del paisaje.

El diagrama radial (Figura 2) muestra cómo el escenario 0, *Situación actual*, aporta los valores más bajos para los servicios de soporte y para la mayor parte de los servicios culturales y de regulación, mientras que para los servicios de abastecimiento aporta valores medios. El escenario 1, *Conservación & Restauración*, presenta valores máximos para todos los servicios culturales, valores más altos para la mayoría de los

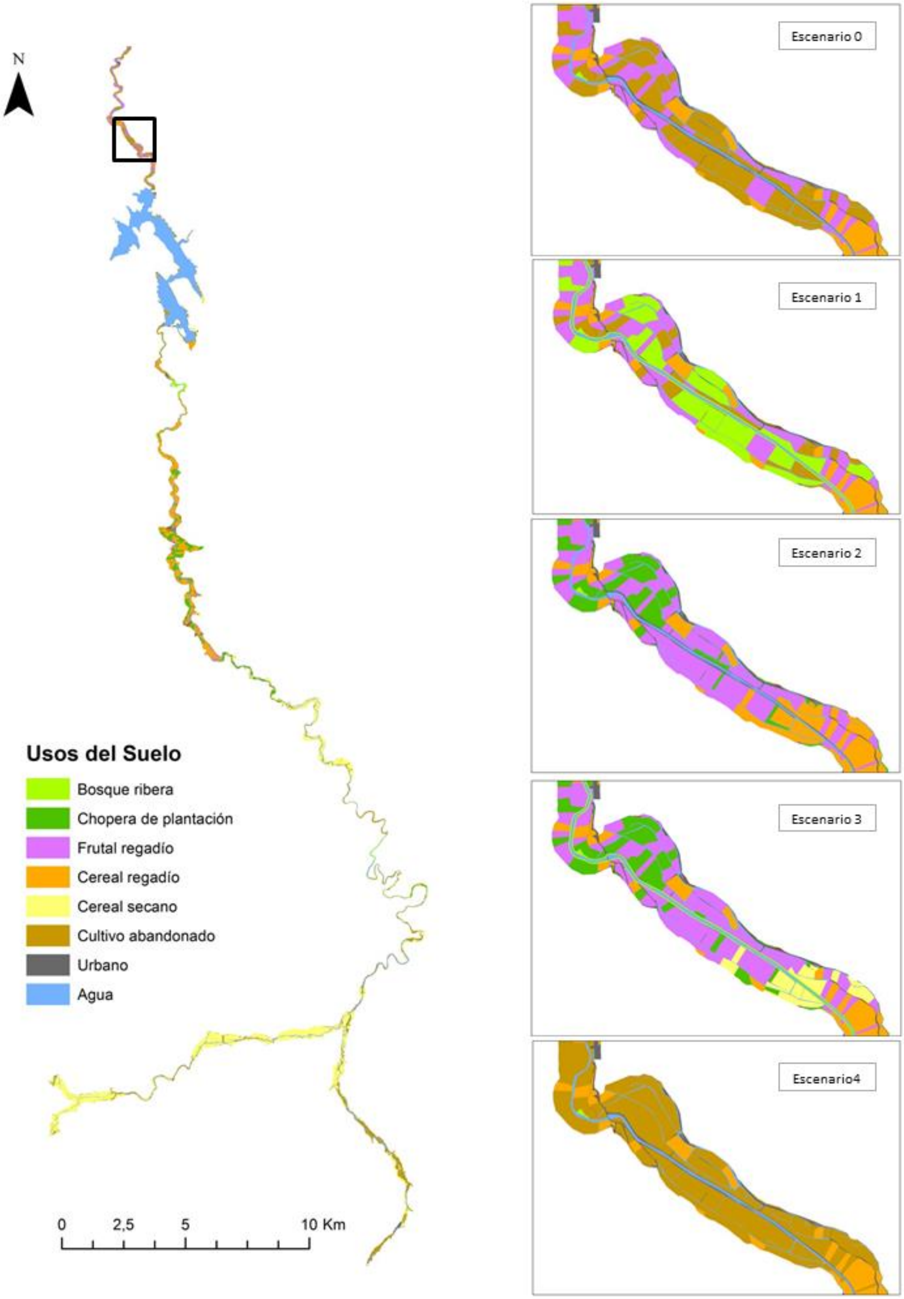


Figura 1. Llanura de inundación del río Piedra (izquierda) y comparación entre escenarios de futuro alternativos (derecha). Escenario 0: *Situación actual*; Escenario 1: *Conservación & Restauración*; Escenario 2: *Agricultura intensiva*; Escenario 3: *Conservación & Producción*; Escenario 4: *Abandono rural*.

indicadores de servicios de soporte y regulación (excepto para formación de suelo, fósforo y regulación de la temperatura), mientras que para los servicios de abastecimiento presenta valores bajos. El escenario 2, *Agricultura intensiva,* presenta los valores mínimos para la mayoría de indicadores de los servicios culturales, de regulación y de soporte (excepto para secuestro de carbono), pero presenta los máximos valores en producción de alimentos. El escenario 3, *Conservación & Producción,* presenta valores intermedios en la mayor parte de servicios de los ecosistemas (aunque son máximos en regulación de la calidad del agua y del clima y mínimo en materias primas). El escenario 4, *Abandono rural,* presenta grandes contrastes ya que aporta valores máximos para algunos indicadores puntuales (formación de suelo, fósforo y materias primas), mientras que los valores son mínimos o muy bajos para el resto de servicios de los ecosistemas.

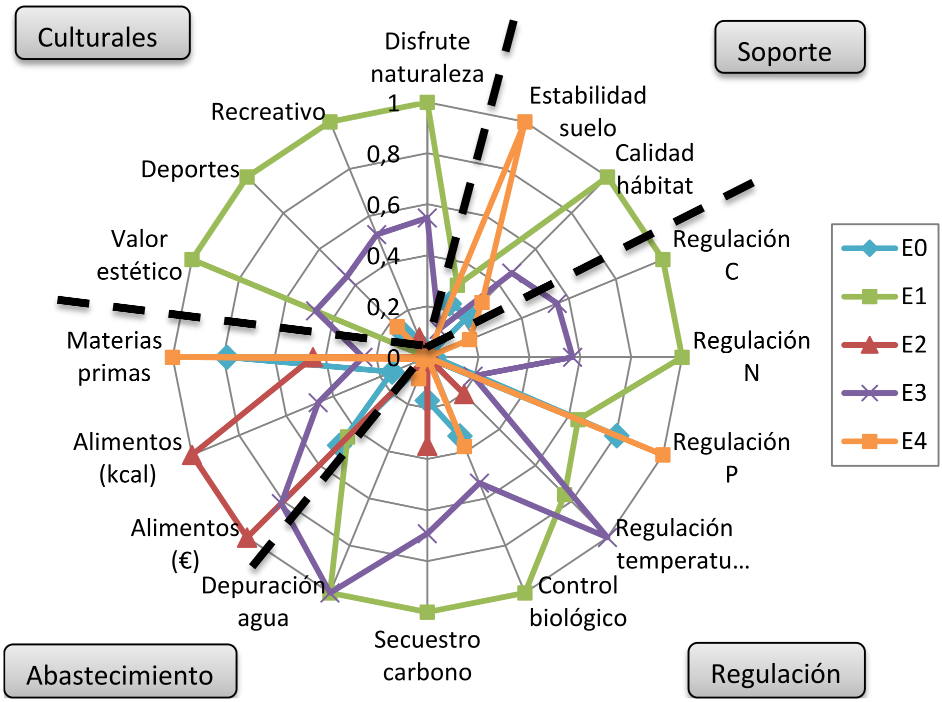


Figura 2. Diagrama radial que representa en cada eje el valor relativo de los servicios de los ecosistemas para cada escenario. Las líneas de colores representan los escenarios: azul (Escenario 0), *Situación actual*; verde (Escenario 1): *Conservación & Restauración*; rojo (Escenario 2): *Agricultura intensiva*; lila (Escenario 3): *Conservación & Producción*; naranja (Escenario 4): *Abandono rural*. Las líneas discontinuas diferencian las categorías de servicios (en sentido horario: servicios de soporte, regulación, abastecimiento y culturales).

Los análisis RDA muestran la relación entre servicios de los ecosistemas en función de los escenarios de futuro (Figura 3). En todos los escenarios, se consigue explicar el 90% de varianza con los tres primeros ejes, donde el primer eje contribuye en más del 50% a la varianza total, y el segundo eje aporta como mínimo otro 25% (Tabla 3). El primer eje separó la depuración del agua del resto de servicios, mientras que el segundo eje separó los servicios de regulación (verde) de los servicios culturales (lila) en todos los escenarios, excepto para el secuestro de carbono, que estuvo agrupado con los servicios culturales en todos los escenarios. Respecto a los municipios, el primer eje se puede relacionar con el tipo de agricultura predominante en los municipios (regadío *vs.* secano), y el segundo con la cobertura de bosque. El escenario 4 muestra el segundo eje de manera invertida al resto de escenarios, pero apoya los resultados comentados.

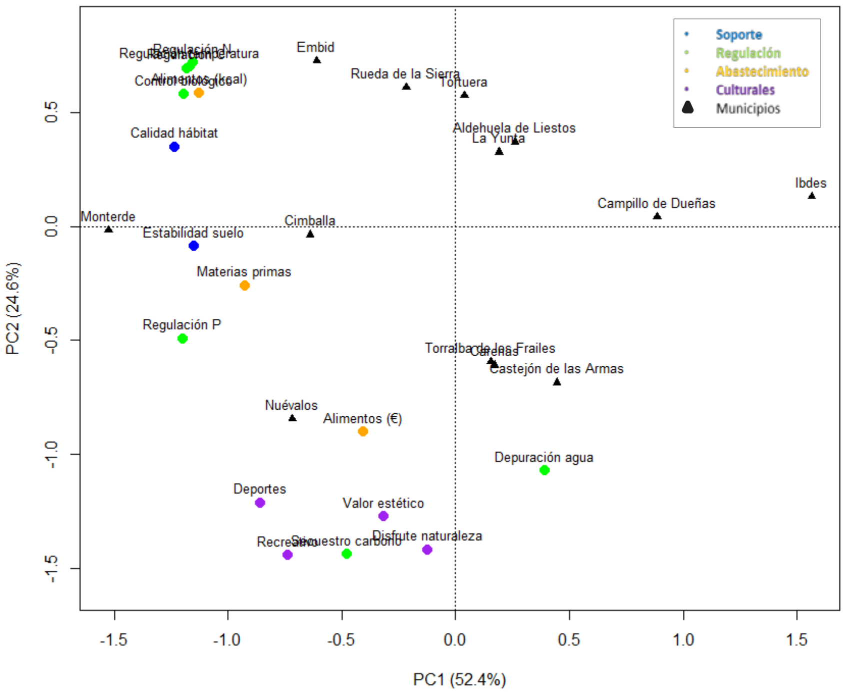
Por otra parte, los resultados del RDA muestran una separación entre servicios de los ecosistemas de manera consistente en los cinco escenarios, indicando la estabilidad de las asociaciones entre servicios de los ecosistemas. Las asociaciones identificadas se han relacionado con actividades concretas. En primer lugar, existe una asociación de servicios de los ecosistemas relacionada con la producción de bienes materiales. Esta asociación se compone de los servicios de producción de alimentos (kcal), regulación del clima (temperatura), regulación del ciclo de nutrientes (C y N), y control biológico de plagas. En segundo lugar, se ha identificado una asociación relacionada con las actividades socio-culturales, que engloba todos los indicadores de servicios culturales y el secuestro de carbono. Además, en el escenario 2 *(Agricultura intensiva)* el indicador económico de la producción de alimentos (€) se asoció con las actividades socio-culturales. Mientras que en los escenarios 1 y 3 *(Conservación & Restauración y Conservación & Producción)*, las materias primas se asociaron con la producción de bienes materiales.

Discusión

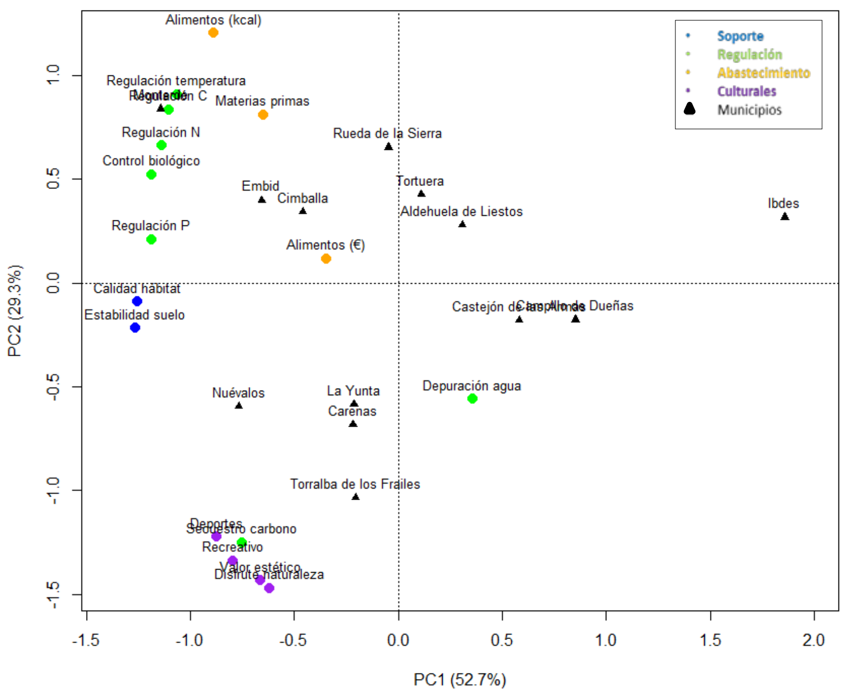
Alternativas para la gestión del valle del río Piedra

En esta modelización de escenarios se han cuantificado los servicios de los ecosistemas proporcionados por la llanura de inundación del río Piedra ante diversas posibilidades de gestión. El resultado más destacable es que todos los escenarios de gestión alternativos a la situación actual permiten incrementar la mayoría de servicios de los ecosistemas y el conjunto de ellos en general; aunque para algunos servicios concretos el escenario *Abandono rural* aporta menor cantidad de servicios. Estos resultados señalan la importancia de incorporar la valoración de los servicios de los ecosistemas a los planes de gestión, pues no considerarlos conlleva a la pérdida de

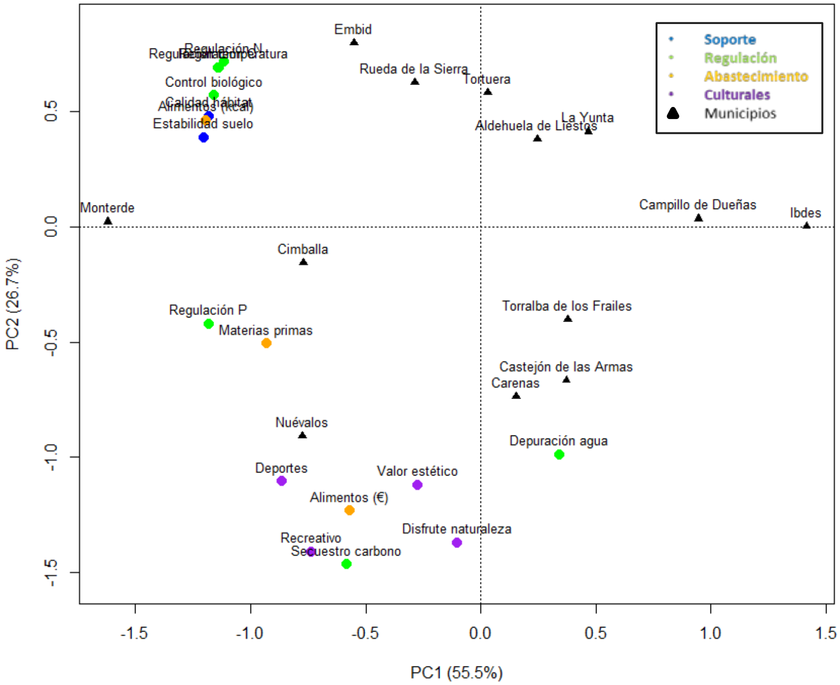
E0 - Situación actual



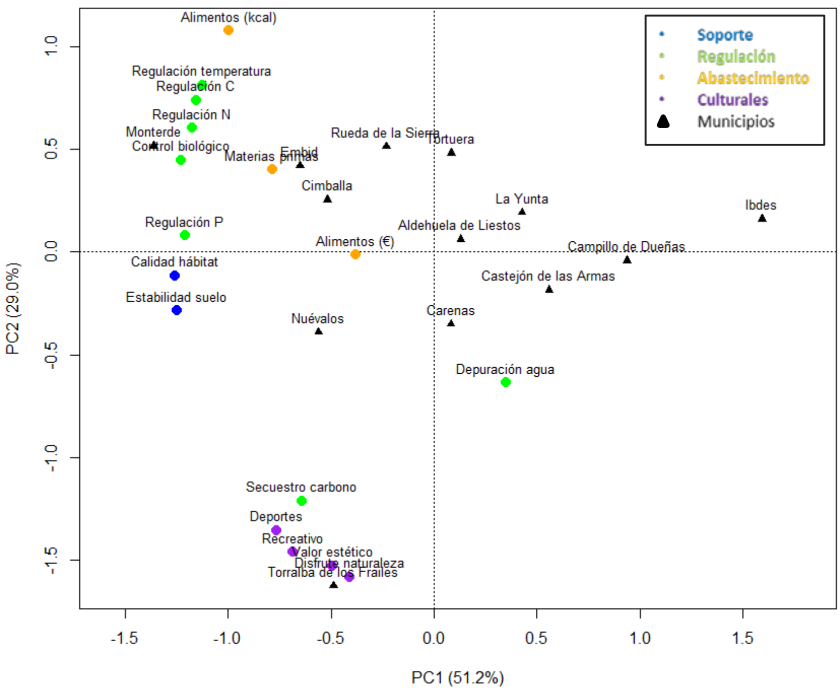
E1 - Conservación & Restauración



E2 - Agricultura intensiva



E3 - Conservación & Producción



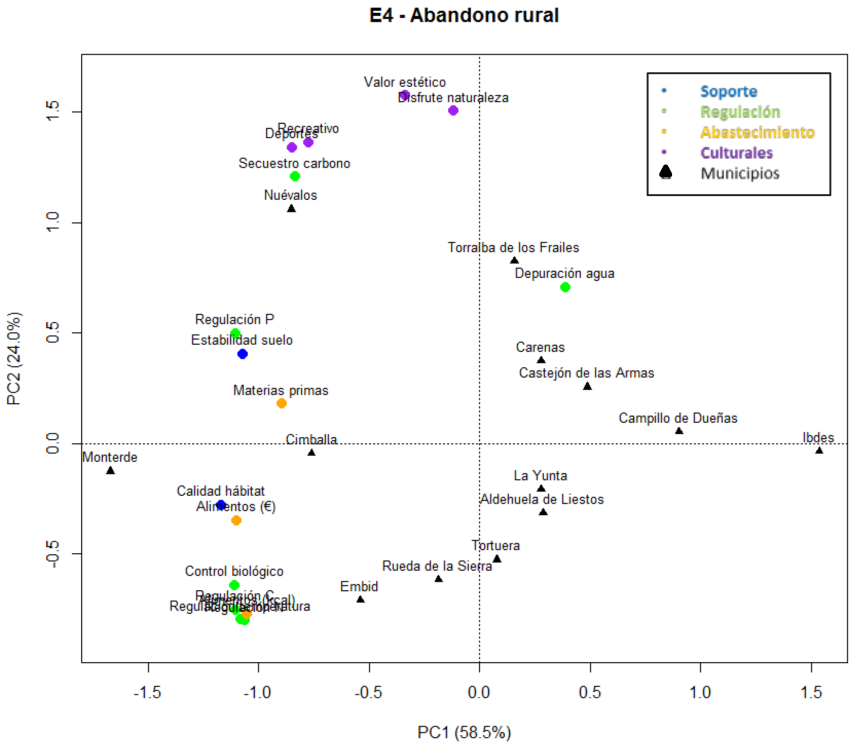


Figura 3. Análisis de Redundancia (RDA) comparativo entre escenarios de futuro. Los círculos indican servicios de los ecosistemas (azul: soporte, verde: regulación, naranja: abastecimiento, lila: culturales) y los triángulos indican los municipios.

servicios para el conjunto de la llanura de inundación (Posthumus *et al.* 2010; Rouquette *et al.* 2011).

Para favorecer el desarrollo del socio-ecosistema basado en el bienestar de la población y la conservación de los recursos naturales y los usos productivos del ecosistema, la selección de alternativas debería depender de varios criterios, como la provisión de servicios de los ecosistemas, los efectos sobre la biodiversidad y la repercusión en las condiciones socio-económicas. Además, sería necesario considerar también la valoración económica y social de los servicios de los ecosistemas e incorporar la participación pública a lo largo de todo el proceso. En este estudio previo, solamente se ha considerado la valoración ecológica de los servicios de los ecosistemas. Por tanto, basándonos en esta perspectiva, la alternativa a escoger dependerá de los servicios que se prefieran favorecer. Para fomentar los servicios de regulación y los culturales, el escenario idóneo sería el de *Conservación & Restauración*. Sin embargo, para fomentar los servicios de abastecimiento el mejor escenario sería el de *Intensificación agrícola*. En el caso de preferir fomentar los

Tabla 3. Estadísticos y puntuaciones para cada servicio de los ecosistemas de los ejes resultantes de los análisis de redundancia (RDA) en cada escenario alternativo de futuro. Escenario 0: *Situación actual*; Escenario 1: *Conservación & Restauración*; Escenario 2, *Agricultura intensiva*; Escenario 3, *Conservación & Producción*; Escenario 4, *Abandono rural*.

	Escenario 0			Escenario 1			Escenario 2			Escenario 3			Escenario 4		
	PC1	**PC2**	**PC3**	**PC1**	**PC2**	**PC3**	**PC1**	**PC2**	**PC3**	**PC1**	**PC2**	**PC3**	**PC1**	**PC2**	**PC3**
Estadísticos															
Eigenvalue	8,385	3,938	2,006	8,430	4,690	1,877	8,887	4,267	1,461	8,194	4,642	2,150	9,360	3,847	0,996
Proporción Explicada	0,524	0,246	0,125	0,527	0,293	0,117	0,555	0,267	0,091	0,512	0,290	0,134	0,585	0,240	0,062
Proporción Acumulada	0,524	0,770	0,896	0,527	0,820	0,937	0,555	0,822	0,913	0,512	0,802	0,937	0,585	0,825	0,888
Servicios de los ecosistemas															
Estabilidad suelo	-0,833	-0,044	0,267	-1,262	-0,219	0,309	-1,203	0,389	0,421	-1,250	-0,285	0,541	-1,068	0,406	0,591
Calidad hábitat	-0,894	0,173	0,090	-1,255	-0,092	0,386	-1,179	0,481	0,318	-1,257	-0,116	0,526	-1,167	-0,279	0,862
Regulación C	-0,854	0,344	-0,055	-1,101	0,835	0,217	-1,143	0,693	-0,012	-1,157	0,739	0,258	-1,101	-0,754	0,527
Regulación N	-0,834	0,356	-0,054	-1,135	0,662	0,290	-1,113	0,716	-0,014	-1,173	0,603	0,326	-1,060	-0,802	0,900
Regulación P	-0,869	-0,244	0,005	-1,187	0,209	-0,704	-1,182	-0,423	-0,417	-1,208	0,080	-0,750	-1,103	0,498	-0,372
Regulación temperatura	-0,845	0,348	-0,084	-1,063	0,907	0,145	-1,135	0,693	-0,113	-1,126	0,813	0,148	-1,082	-0,798	0,632
Control biológico	-0,864	0,288	-0,049	-1,188	0,521	0,184	-1,159	0,572	-0,139	-1,227	0,447	0,166	-1,110	-0,642	0,811
Secuestro carbono	-0,347	-0,715	-0,472	-0,752	-1,252	-0,910	-0,582	-1,468	-0,990	-0,640	-1,211	-1,271	-0,835	1,208	-0,967
Depuración agua	0,283	-0,533	-0,517	0,360	-0,562	-2,042	0,345	-0,990	-1,433	0,349	-0,636	-1,634	0,388	0,709	0,541
Alimentos (€)	-0,292	-0,448	-0,746	-0,345	0,117	-2,550	-0,571	-1,235	-1,632	-0,381	-0,014	-2,357	-1,099	-0,347	-1,028
Alimentos (kcal)	-0,816	0,292	-0,155	-0,887	1,205	-0,139	-1,195	0,465	-0,066	-0,999	1,077	-0,064	-1,051	-0,772	-0,256
Materias primas	-0,670	-0,129	-0,212	-0,646	0,810	-1,040	-0,931	-0,507	-0,866	-0,787	0,399	-1,351	-0,893	0,180	-2,252
Valor estético	-0,229	-0,632	0,564	-0,664	-1,435	0,480	-0,272	-1,123	2,162	-0,495	-1,526	0,685	-0,337	1,575	0,961
Deportes	-0,619	-0,603	0,296	-0,875	-1,225	0,124	-0,863	-1,107	0,892	-0,767	-1,355	0,255	-0,847	1,340	-0,276
Recreativo	-0,533	-0,717	0,114	-0,794	-1,341	0,058	-0,734	-1,412	0,487	-0,688	-1,460	0,041	-0,775	1,363	-0,228
Disfrute naturaleza	-0,090	-0,705	0,331	-0,621	-1,471	0,278	-0,102	-1,373	1,100	-0,411	-1,581	0,544	-0,118	1,505	1,417

servicios de soporte no está claro cuál es el escenario más conveniente, ya que el escenario de *Abandono rural* presenta el valor máximo para un indicador y el escenario de *Conservación & Restauración* para el otro, por lo que habría que analizar más indicadores de servicios de soporte.

En general, se aprecia que el escenario de *Conservación & Restauración* maximiza la mayoría de servicios de los ecosistemas. Sin embargo, este escenario apenas provee servicios de abastecimiento, fundamentales para mantener la población local en el valle del Piedra pues la mayoría de los habitantes son agricultores. Por ello, podría sugerirse como más apropiado un escenario intermedio que proporcione todos los servicios de los ecosistemas necesarios para mantener el flujo de servicios y la continuidad del socio-ecosistema. En este caso, el escenario de *Conservación & Producción*, proporciona un conjunto de servicios de manera equilibrada, ya que combina la conservación y mejora de los servicios culturales, de soporte y de regulación asociados al escenario *Conservación & Restauración*, y los beneficios derivados de la producción agrícola asociados al escenario *Agricultura intensiva*, maximizando la depuración del agua, clave para el mantenimiento de otros servicios de los ecosistemas intermedios y finales.

Conseguir un conjunto equilibrado de servicios de los ecosistemas es una de las principales justificaciones que apoyan la conservación de paisajes multifuncionales, es decir, de aquellos paisajes que desempeñan múltiples funciones en el socio-ecosistema. Sin embargo, otros modelos de gestión prefieren separar las funciones del paisaje para aumentar su eficiencia, dedicando unas zonas a la producción intensiva de alimentos y otras exclusivamente a la conservación de la biodiversidad. Este debate se conoce como el dilema entre "*land sharing*" y "*land sparing*" (Balmford *et al.* 2012), es decir, entre usos compartidos frente usos separados del territorio, que lleva años polarizándose sin llegar a resolverse (Phalan *et al.* 2011; Tscharntke *et al.* 2012; Scariot 2013). Para avanzar en el debate, recientemente se ha propuesto valorar los paisajes agrícolas como sistemas socio-ecológicos e identificar las propiedades de estos sistemas que permiten compaginar la seguridad alimentaria con la biodiversidad (Fischer *et al.* 2014). En el valle del Piedra, esto podría lograrse implementando un modelo de gestión similar al de *Conservación & Producción* propuesto. En este caso, es necesario tener en cuenta que la producción agrícola de este valle apenas contribuye a la seguridad alimentaria de sus habitantes, sino que actúa como fuente de ingresos. Por ello, los agricultores y el resto de agentes sociales de interés deberían participar en la toma de decisiones sobre las estrategias de gestión para satisfacer el mayor número de objetivos propuestos. Además, existen otras posibilidades de gestión que podrían contribuir a compaginar la seguridad alimentaria con la biodiversidad, como la agricultura ecológica, la conservación de linderos, y otras medidas de verdeo ("*greening*") propuestas por la nueva PAC (Política Agraria Común 2014 – 2020). Estos escenarios intermedios son interesantes de analizar pero menos aplicables al objetivo demostrativo de este capítulo por la escasez de referencias sobre los efectos específicos en cada servicio de los ecosistemas. La comparación entre los cinco modelos de escenarios de futuro propuestos (cuatro escenarios alternativos frente a la situación actual) ha permitido mostrar el efecto de distintas alternativas de gestión en la provisión de servicios de los ecosistemas a escala de paisaje y la necesidad de actuar en el valle del Piedra para evitar la pérdida generalizada de servicios. La escasez en la provisión de servicios en esta zona, como ocurre en otras zonas rurales en España, es derivada de dos hechos opuestos: la intensificación de la agricultura en las mejores tierras para el cultivo y el abandono de las restantes (García-Llorente *et al.* 2012).

En cualquier caso, la gestión de los ecosistemas debe hacerse considerando, efectivamente, a los ecosistemas como sistemas y no únicamente como fuente de recursos. Además, deberíamos reconocer los límites de los ecosistemas para proveer servicios (Rockström *et al.* 2009) y definir las necesidades humanas reales para sentar las bases de un modo de vida sostenible, basado en una gestión equilibrada entre las demandas sociales y la capacidad de los ecosistemas para proporcionar servicios. Por ello, una gestión adaptativa a las condiciones cambiantes del entorno y de la sociedad, así como una gobernanza a varios niveles políticos, basada en instituciones ágiles y abiertas a la participación de su público, es fundamental para desarrollar el aprendizaje y la adaptación necesaria para el buen funcionamiento de todos los componentes del socio-ecosistema.

Identificando asociaciones de servicios de los ecosistemas

Se ha sugerido que los servicios de los ecosistemas aparecen agrupados entre sí (Bennett *et al.* 2009; Raudsepp-Hearne *et al.* 2010). En la llanura de inundación del río Piedra, se ha observado una asociación relacionada con la producción de bienes materiales y otra con las actividades socio-culturales. Estos resultados reflejan la intrínseca dependencia entre servicios de los ecosistemas de diferentes categorías, especialmente evidente para la asociación relacionada con los bienes materiales. En esta asociación se observa la importancia de conservar los servicios de regulación para obtener bienes materiales para el ser humano. En consecuencia, justifica las políticas de gestión que fomentan la conservación de un conjunto de servicios de los ecosistemas en lugar de la dominancia de unos servicios sobre otros. Por otra parte, la elevada correlación entre los indicadores de servicios de regulación y entre los servicios culturales facilita la gestión de estos servicios, pues indica que las medidas destinadas a favorecer un servicio de regulación o uno cultural favorecerán a sus respectivos grupos en conjunto.

Los resultados mostraron también de manera consistente una clara oposición en la provisión de servicios de los ecosistemas entre municipios cuya llanura de inundación conserva el bosque de ribera frente a los que está dedicada fundamentalmente a la agricultura, y dentro de ésta, la dedicada al secano frente al regadío. Esta oposición podría reducirse aplicando medidas para favorecer el conjunto de servicios de los ecosistemas de manera equilibrada, como las señaladas en el escenario *Conservación & Producción* y las planteadas en el capítulo 3 (Felipe-Lucia and Comín 2015) y el capítulo 4 (Felipe-Lucia *et al.* 2014a). Concretamente, estas medidas deberían mejorar la provisión diversificada de servicios de los ecosistemas en los usos del suelo que ocupan mayor extensión y reducir los antagonismos entre servicios de los ecosistemas. Dichas medidas podrían llevarse a cabo mediante podas controladas en frutales y bosques de ribera, reduciendo el uso de fertilizantes y pesticidas en la agricultura, conservando los linderos, manteniendo los cultivos productivos, conservando una franja mínima de cinco metros de bosque de ribera en las tierras de cultivo e incrementando la superficie del bosque de ribera. Además, sería necesario implementar medidas específicas que fomenten los servicios de los ecosistemas deseables pero con valores bajos en el escenario elegido. En el caso del escenario *Conservación & Producción*, la estabilidad del suelo y la regulación del fósforo, podrían mejorarse mediante medidas de conservación del suelo tales como la agricultura de conservación (Pretty 2008), que evita el laboreo de los barbechos y frutales.

En conclusión, los resultados de este trabajo constituyen una base para la elaboración de escenarios de futuro en el valle del Piedra de manera participativa. Los análisis realizados han permitido demostrar que la valoración de los servicios de los ecosistemas constituye una herramienta útil para gestionar los ecosistemas de manera sostenible porque incorpora factores biofísicos y sociales. Los resultados también señalan la importancia de incorporar la valoración de los servicios de los ecosistemas a los planes de gestión para evitar la pérdida generalizada de servicios. Mediante un escenario de conservación y producción intermedio, es posible favorecer el desarrollo del socio-ecosistema basado en el bienestar de la población y la conservación de los recursos naturales y los usos productivos del ecosistema. Este trabajo podría complementarse incorporando el análisis de las formas de gobernanza de cada servicio de los ecosistemas para identificar las formas de gobernanza, que en combinación con los escenarios alternativos de uso del suelo, proporcionan el acceso a los servicios de los ecosistemas a un mayor número de agentes sociales de interés.

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CAPÍTULO 8. DISCUSIÓN GENERAL

Aportaciones de esta tesis doctoral

Esta tesis doctoral contribuye a interpretar la relación entre los aspectos ecológicos y sociales que influyen en el flujo de servicios de los ecosistemas y a aplicar el análisis de estas interacciones a la gestión de los ecosistemas. Por una parte, en esta tesis doctoral se analizan las relaciones entre los servicios que aportan los principales usos del suelo identificados en la llanura de inundación del río Piedra y la biodiversidad vegetal asociada a ellos (Capítulo 3). Además, se han identificado las sinergias y antagonismos entre servicios de los ecosistemas en los diferentes usos del suelo y a tres escalas espaciales: parcela, municipio y paisaje (la llanura de inundación en conjunto) y se ha propuesto una clasificación de las interacciones entre servicios de los ecosistemas que incorpora los valores sociales que rigen las decisiones de gestión, junto con los factores biofísicos, como posibles causas de la existencia de antagonismos entre servicios de los ecosistemas (Capítulo 4). Por otra parte, se han desarrollado unas directrices que sugieren los puntos fundamentales a incluir en la valoración social de los servicios de los ecosistemas para que los resultados puedan ser comparables y transferibles, y se ha aplicado el modelo propuesto en la valoración social de los servicios de los ecosistemas del valle del río Piedra (Capítulo 5). Además, se han explorado las interacciones tanto ecológicas como sociales que intervienen en el flujo de servicios de los ecosistemas al bienestar humano, identificando los servicios clave para mantener el flujo de servicios de los ecosistemas y las asimetrías de poder entre agentes de interés que determinan el acceso y la gestión de los servicios de los ecosistemas (Capítulo 6). Por último se han comparado cinco escenarios alternativos de gestión de la llanura de inundación del río Piedra en busca de una combinación más equilibrada de servicios (Capítulo 7).

Las aportaciones de esta tesis doctoral son útiles para integrar el conjunto de servicios que proporcionan los ecosistemas en políticas ambientales y territoriales que valoren los servicios desde múltiples perspectivas, fomenten los paisajes multifuncionales, suministren un conjunto equilibrado de servicios, incluyan la participación pública y aseguren un acceso igualitario a los servicios de los ecosistemas. En concreto, esta tesis contribuye en diferentes campos de investigación con las aportaciones que se discuten a continuación.

Oportunidades de las llanuras de inundación para la provisión de múltiples servicios de los ecosistemas

Una de las principales aportaciones de esta tesis doctoral es ampliar el conocimiento sobre los múltiples servicios que las llanuras de inundación dominadas por agroecosistemas pueden ofrecer mientras conservan la biodiversidad asociada a ellas (Capítulo 3). Los paisajes multifuncionales, como los agroecosistemas, proporcionan diferentes tipos de servicios (Bennett and Balvanera 2007; Lovell and Johnston 2008) y han sido gestionados con éxito para asegurar su rentabilidad y mejorar la provisión de servicios de los ecosistemas en numerosos casos (de Groot 2006; Fischer *et al.* 2006; Scherr and McNeely 2008; Anton *et al.* 2010). Se ha observado que cada tipo de uso del suelo proporciona los distintos servicios de los ecosistemas con diferente valor (Posthumus *et al.* 2010) debido a la diferente estructura biológica y física de cada uso y a las diferentes funciones que éstas ejercen en el ecosistema (de Groot *et al.* 2010; Isbell *et al.* 2011). Por ello, es posible fomentar de forma combinada la conservación de la biodiversidad y los servicios de los ecosistemas mediante la gestión adecuada de cada uso del suelo, pero especialmente mediante la gestión integral del conjunto de la llanura de inundación.

Nuestros resultados coinciden con estudios recientes (p.ej. Harrison *et al.* 2010; Petz and van Oudenhoven 2012) en que los usos del suelo cultivados (es decir, los cultivos de cereal de regadío y secano, los frutales y las choperas de plantación) básicamente proporcionan servicios de abastecimiento, mientras que los hábitats naturales o semi-naturales proveen la mayor parte de servicios de los ecosistemas pero apenas de abastecimiento (Capítulos 3 y 4). Además, los resultados de esta tesis doctoral apoyan los esfuerzos científicos e institucionales para promover políticas que conecten la conservación de la biodiversidad con la mejora de la provisión de servicios de los ecosistemas (Díaz *et al.* 2006; Chan *et al.* 2006; Turner *et al.* 2007; Palumbi *et al.* 2008; Bello *et al.* 2010), pues muestran la relación existente entre una elevada provisión de servicios de los ecosistemas y mayor conservación de la biodiversidad (Benayas *et al.* 2009).

Por otra parte, se han identificado los aspectos de la biodiversidad que más se correlacionan con cada servicio de los ecosistemas usando varios índices de diversidad vegetal (Capítulo 3). El alto grado de correlaciones positivas encontradas confirma las sinergias entre biodiversidad y servicios de los ecosistemas, especialmente evidentes para los servicios culturales y de soporte. Por otra parte, también encontramos un antagonismo directo entre los servicios de provisión de alimentos y los índices de abundancia de diversidad vegetal. Además, a pesar de que la mayoría de los estudios que relacionan los servicios de los ecosistemas con la biodiversidad basan sus resultados únicamente en el análisis de la riqueza de especies (Balvanera *et al.* 2006), nuestros resultados muestran que las correlaciones entre servicios de los ecosistemas y biodiversidad varían en función del indicador utilizado para estimar la biodiversidad. Por ejemplo, la regulación de gases (secuestro de carbono) mostró una elevada correlación con los índices de riqueza y diversidad de formas de crecimiento, pero no con el resto de índices de diversidad vegetal. Además, las interacciones entre servicios de los ecosistemas y biodiversidad variaron en los diferentes usos del suelo. De manera que valorar los servicios de los ecosistemas y la biodiversidad en cada uso del suelo por separado y considerar varias medidas de la biodiversidad permite alcanzar una mejor comprensión del funcionamiento de los ecosistemas, fundamental para diseñar políticas de gestión que mejoren al mismo tiempo los servicios de los ecosistemas y la conservación de la biodiversidad.

No obstante, la comunidad científica continúa debatiendo sobre la capacidad real de implementar políticas que beneficien tanto la provisión de servicios de los ecosistemas como la conservación de la biodiversidad (Redford and Adams 2009; Skroch and López-Hoffman 2010; Reyers *et al.* 2012b; Faith 2012). Las dificultades para encontrar soluciones conjuntas se incrementan cuando se tratan de implementar políticas a gran escala (Chan *et al.* 2006; Naidoo *et al.* 2008), pero pueden reducirse si las políticas se diseñan y se implementan a menor escala (Turner *et al.* 2007). De hecho, la escala de análisis puede cambiar la estimación de los servicios de los ecosistemas proporcionados (Hein *et al.* 2006), ya que el cambio en la proporción de los usos del suelo a diferentes escalas influye en el tipo y cantidad de servicios de los ecosistemas provistos a cada una (Capítulo 4). Por ejemplo, nuestros resultados demuestran que a escala de parcela, los bosques de ribera proporcionan mayor cantidad de servicios de los ecosistemas, mientras que a escala municipal y de paisaje los cultivos de cereal proporcionan mayor cantidad de servicios debido a que ocupan la mayor extensión de terreno. Por tanto, las políticas de gestión deberían ser adaptativas, es decir, gestionando las llanuras de inundación a escala de paisaje, pero siendo capaces de adaptar medidas específicas para cada tipo de uso del suelo.

La escala espacial de análisis también condiciona el alcance de las interacciones entre servicios de los ecosistemas. En el área estudiada, sólo cuatro interacciones entre servicios de los ecosistemas permanecieron constantes a las tres escalas espaciales consideradas (parcela, municipio y paisaje), destacando la estabilidad de ciertas interacciones (sinergias entre estabilidad del suelo, regulación de nutrientes y calidad del hábitat). Sin embargo, la mayoría de las interacciones variaron a diferentes escalas espaciales, indicando que no hay una única escala relevante para analizar las interacciones entre servicios de los ecosistemas y justificando la necesidad de conocer cómo responden los servicios a cada escala espacial, para una gestión eficiente de los servicios de los ecosistemas. Además, estos análisis han permitido distinguir el uso del suelo y la escala espacial en la que tienen lugar las correlaciones, destacando el tipo de cobertura y uso del suelo como factor esencial que controla dichas interacciones.

Por otro lado, la gestión de los ecosistemas para fomentar ciertos servicios suele conllevar la pérdida de otros, aparentemente por incompatibilidad o antagonismo (*trade-off*). Los antagonismos en la provisión de servicios de los ecosistemas se pueden clasificar en función de la escala espacial y temporal a la que actúen y según la reversibilidad del servicio comprometido (Millennium Ecosystem Assessment 2005). Aunque estos antagonismos derivan de las medidas de gestión implementadas, existe poca investigación sobre los valores y preferencias sociales que subyacen detrás de estas decisiones de gestión. Por ello, se han clasificado los antagonismos entre servicios de los ecosistemas según sean causados por factores ecológicos o por factores sociales (Capítulo 4, tabla 7). En el primer caso, las interacciones entre servicios de los ecosistemas pueden ser consistentes para todos los usos del suelo (p.ej. la producción de alimentos es incompatible con la de materias primas) o deberse precisamente al tipo de uso de suelo (p.ej. en zonas urbanas no existe formación de suelo). En el segundo caso, el compromiso se debe a las decisiones de gestión derivadas de preferencias y valores sociales (p.ej. la provisión de alimentos no se compatibiliza con el espacio recreativo). Conocer las causas que originan la pérdida de unos servicios de los ecosistemas en favor de otros es fundamental para lograr una gestión que minimice estas pérdidas (Howe *et al.* 2014).

Los agroecosistemas ocupan alrededor del 40% de la superficie terrestre (FAO 2009), de la cual el 3,5% se sitúa en llanuras de inundación (Tockner and Stanford 2002), homogeneizando el paisaje. Fomentar los modelos de gestión destinados únicamente a proveer servicios de abastecimiento es insostenible, ya que esta gestión limita la provisión del resto de servicios de los ecosistemas. Además, esa gestión suele ir asociada a una profunda alteración de los cauces fluviales, que a largo plazo puede poner en riesgo los propios servicios de abastecimiento por la desregulación del flujo de caudales. En las llanuras de inundación dominadas por agroecosistemas, que son frecuentes en zonas rurales de Europa y Norte América, es preferible apostar por un mosaico multifuncional de hábitats (Reyers *et al.* 2012a; Mitchell *et al.* 2014) compuesto de cultivos productivos, choperas y frutales que conserven un cinturón forestal a lo largo de las riberas (Srivastava and Vellend 2005), además de bosques de ribera restaurados (Luck *et al.* 2009; Meli *et al.* 2014). Este modelo de gestión permitiría frenar la pérdida de biodiversidad (Billeter *et al.* 2008), reducir los "*dis-ervicios*[1]" existentes (Power 2010) y revertir los causados en el pasado (Swinton *et al.* 2007), incrementando la provisión de servicios y la resiliencia de los ecosistemas, minimizando las pérdidas en servicios de abastecimiento (Polasky *et al.* 2005) y creando sinergias para los servicios culturales. De este modo, se podría impulsar el turismo rural, conservar las especies, razas y variedades locales, promocionar los productos locales, crear puestos de trabajo y contribuir a evitar la despoblación del medio rural.

Contribuciones para la valoración social de los servicios de los ecosistemas

Esta tesis doctoral contribuye a impulsar la valoración de los servicios de los ecosistemas desde el punto de vista social proponiendo unas directrices que permiten comparaciones entre estudios y sirven para avanzar en el conocimiento de los valores que la sociedad otorga a los ecosistemas y mejorar los planes de gestión basados en el fomento de los servicios de los ecosistemas (Capítulo 5). Las directrices consisten en identificar en tales evaluaciones tres aspectos básicos: a) el contexto espacial y temporal, b) el contexto social (quién participa en la valoración) y c) la metodología empleada para valorar los servicios.

Respecto del contexto espacial, la revisión bibliográfica efectuada ha constatado que la mayoría de los estudios se han llevado a cabo a escala municipal o supra local, y que la valoración social de los servicios de los ecosistemas se está llevando a cabo mayoritariamente en ecosistemas forestales, cultivados, y humedales, es decir, en los mismos ecosistemas que se evalúan desde las perspectivas ecológica y económica (Feld *et al.* 2009; Martin *et al.* 2012). La valoración social de los servicios de algunos tipos de ecosistemas (p.ej. las zonas desérticas o polares) apenas se conoce; sin embargo, considerar los valores que las poblaciones de estos lugares otorgan a los ecosistemas permitiría ampliar nuestra percepción actual sobre el valor de los ecosistemas y mejorar los proyectos de gestión del territorio y de los recursos naturales en estas y otras áreas.

Los resultados del contexto social muestran que aunque los residentes locales son el grupo más representado, son incluidos en poco más de un tercio de las evaluaciones sociales de servicios de los ecosistemas. Considerar a los agentes locales de interés en la valoración de servicios de los ecosistemas e incorporar sus opiniones y preocupaciones en los planes de gestión que de éstos se deriven puede contribuir al éxito de dichos proyectos (Hicks *et al.* 2009; Moreno *et al.* 2014). Sin embargo, la selección de participantes y los métodos empleados para legitimar los resultados obtenidos debe hacerse cuidadosamente, ya que la estructura social y cultural de las poblaciones locales son relevantes a la hora de identificar y valorar diferentes ecosistemas y servicios (Martín-López *et al.* 2012; Iniesta-Arandia *et al.* 2014). No obstante, rechazar las opiniones locales puede reducir el éxito de proyectos que pretenden mejorar la provisión de servicios de los ecosistemas si dichos servicios no son apoyados localmente (Comín *et al.* 2005; Hauck *et al.* 2013). De igual modo, cuestiones surgidas al nivel local tienen más posibilidades de ser implementadas si éstas implican a gestores de las esferas donde tiene lugar la toma de decisiones.

Respecto de los métodos de valoración, los dos métodos más utilizados (identificación de servicios de los ecosistemas y orden de preferencias) se consideran idóneos cuando se aplican de manera combinada (Tallis *et al.* 2012; Ringold *et al.* 2013); es decir, identificando en primer lugar los servicios de los ecosistemas valiosos para los agentes de interés y, en segundo lugar, ordenando sus preferencias. Este doble método es especialmente importante para legitimar las políticas de gestión, donde suele haber antagonismos entre servicios de los ecosistemas derivados de favorecer unos usos del suelo en lugar de otros (Hicks *et al.* 2013). Además, en esta tesis doctoral se ha señalado la importancia de distinguir claramente la valoración social de los servicios de los ecosistemas de la valoración económica basada en preferencias sociales, ya que en la mayor parte de los estudios revisados ambos conceptos aparecen mezclados. La valoración económica de los servicios de los ecosistemas basada en preferencias sociales sigue siendo la más utilizada, pese a que los resultados pueden estar sesgados por la condición económica de los encuestados (Gómez-Baggethun and Ruiz-Pérez 2011; Costanza *et al.* 2014), entre otros muchos riesgos (Funtowicz and Ravetz 1994; Chee 2004; Wegner and Pascual 2011; Farley 2012; Casado-Arzuaga *et al.* 2013; Villa *et al.* 2014a). Para fomentar la valoración social frente a la valoración económica como método para evaluar los servicios de los ecosistemas por parte de la sociedad es necesario explicitar los métodos empleados; concretamente, el contexto espacio-temporal, el contexto social y la metodología. Esta tesis doctoral presenta un marco conceptual para explicitar estos tres aspectos y aplicar empíricamente la integración de dicha información en la valoración social de los servicios de los ecosistemas. No obstante, utilizar un enfoque integrado de la valoración ecológica, social y económica de los servicios de los ecosistemas proporciona información más detallada para asesorar la gestión de los ecosistemas, basadas en criterios de sostenibilidad ecológica, eficiencia económica y equidad (Costanza 2000; Millennium Ecosystem Assessment 2005; Farley 2012), especialmente relevante cuando se crean antagonismos por la provisión de unos servicios en favor de otros.

Aplicaciones del análisis de las relaciones de poder en el estudio de los flujos de servicios de los ecosistemas

Esta tesis doctoral contribuye a identificar la existencia de las relaciones de poder que modulan tanto la provisión de servicios de los ecosistemas como las interacciones entre los grupos de interés de los socio-ecosistemas. Integrar las interacciones ecológicas y sociales a lo largo del flujo de servicios de los ecosistemas es clave para comprender las posibles asimetrías entre grupos sociales de interés, derivadas de las políticas de gestión ambiental, y para promocionar una gestión sostenible de los servicios de los ecosistemas.

Se ha demostrado que el marco conceptual propuesto (Capítulo 6, figura 1), basado en el modelo de cascada[2], es útil para identificar servicios de los ecosistemas claves que determinan la provisión de otros servicios y servicios de los ecosistemas importantes para cada grupo de interés. De esta manera, se ha determinado la capacidad de cada grupo de interés de gestionar los servicios de los ecosistemas que utilizan, su implicación en la provisión y uso de otros servicios y las asimetrías de poder entre grupos de interés derivadas de éstas.

Los análisis realizados permitieron detectar qué servicios de los ecosistemas son gestionados por un único grupo de interés, destacando el poder de dicho grupo para controlar el acceso y uso de los servicios de los ecosistemas. En nuestro caso de estudio, se identificaron cuatro grupos de interés: sector primario, sector recreativo, de ocio, e instituciones (Capítulo 6, Tabla 2). Las instituciones fueron el grupo con mayor poder, ya que cuentan con la capacidad de gestionar los servicios de soporte y regulación clave para la existencia de otros servicios de los ecosistemas y que son utilizados por la mayor parte de los grupos de interés. Además, este grupo tiene el poder de promover sinergias o antagonismos entre servicios de los ecosistemas y de limitar el uso de los servicios que gestionan a agentes de interés concretos, potencialmente creando desequilibrios sociales. El sector recreativo cuenta con bastante poder ya que gestionan y utilizan los servicios culturales que dirigen la economía de la zona. Finalmente, el grupo de ocio y en parte el sector primario fueron los más vulnerables ya que tienen escasa capacidad de gestión sobre los servicios de los ecosistemas que utilizan. El sector primario tiene cierta influencia sobre los servicios de provisión y además, algunas de sus prácticas deterioran servicios críticos para la integridad de otros servicios de los ecosistemas, afectando a los usuarios dependientes de dichos servicios.

Nuestros análisis han permitido distinguir cómo el sistema de gobernanza de cada servicio de los ecosistemas puede condicionar el acceso a los servicios. Por ejemplo, sistemas de gestión basados en un único grupo de interés en el que los servicios de los ecosistema son usados y gestionados por este grupo puede generar un efecto de retroalimentación positiva, causando dos resultados opuestos: reforzar el servicio (por ejemplo, las actividades recreativas atraen más actividades), o agotarlo (por ejemplo, la fertilidad natural del suelo desaparece al consumirse a tasas mayores que su regeneración). A pesar de que la gestión basada en un único grupo de interés debería conducir a la auto-regulación del servicio (retroalimentación negativa), existe un alto riesgo de entrar en un círculo vicioso en el que el servicio es consumido por un uso no regulado del mismo, disminuyendo la capacidad del sistema para proveer servicios.

Por otra parte, se ha identificado que el sistema de gestión existente en el valle del Piedra es "*de arriba abajo*", es decir, la gestión se dirige desde los niveles más altos de la administración –implicando normalmente a agentes sociales de interés externos al sistema– hacia la población local. Este sistema de gestión no fomenta las potenciales sinergias entre los servicios de los ecosistemas generados en esta área, como la calidad del hábitat y los servicios culturales (Capítulo 4), ni fortalece la gobernanza de las comunidades sobre sus recursos. Sin embargo, ejemplos de sistema de gestión participativos "*de abajo a arriba*", como la gestión descentralizada de bosques en Tanzania (Lund and Treue 2008), de ecosistemas costeros en Kenia (Forrester *et al.* 2014) y de estuarios en Sudáfrica (Bowd *et al.* 2012), han demostrado ser importantes complementos de los sistemas "*de arriba a abajo*" existentes. Estos sistemas de gestión participativos reconectan el círculo del sistema de gobernanza "*de arriba a abajo*" y viceversa, mejoran el conocimiento ecológico de los agentes de interés y potencian un acceso más democrático a los servicios de los ecosistemas.

Perspectivas para futuras líneas de investigación

En este apartado se describen brevemente algunas líneas de investigación con potencial para su desarrollo. Estas líneas de investigación derivan directamente de los resultados de esta tesis doctoral o están relacionados con ellos y durante la elaboración de este trabajo se han observado como de interés para continuar la investigación en servicios de los ecosistemas.

Desarrollar indicadores para evaluar los servicios de los ecosistemas en diferentes contextos de manera precisa. Aunque ya existen varios proyectos que han investigado en este campo (p.ej. CICES – Common International Classification of Ecosystem Services – de la Agencia Medioambiental Europea, ESID – Ecosystem Service Indicator Database – del World Resources Institute), todavía es necesario conocer cuáles son los mejores indicadores para evaluar cada ecosistema (Müller *et al.* 2006; van Oudenhoven *et al.* 2012), cuáles son más sensibles a cada tipo de uso del suelo, y cómo aplicar e interpretar los indicadores. Para ello, sería necesario colaborar internacionalmente para recopilar datos de diferentes tipos de ecosistemas y usos de suelo, y contrastar los datos obtenidos localmente con la información derivada de métodos SIG y teledetección a diferentes escalas espaciales. Este campo de investigación expandiría el potencial de los servicios de los ecosistemas para asesorar en políticas ambientales específicas para cada uso del suelo y ecosistema.

Identificar grupos de servicios de los ecosistemas compatibles y sinérgicos. Por ejemplo, continuando con el sub-programa PECS (Programme for Ecosystem Change and Society) que trata de establecer grupos de servicios de los ecosistemas (“*bundles*”) que aparecen juntos en diferentes casos de estudio. Esta iniciativa podría complementarse tratando de identificar también si existen grupos de servicios de los ecosistemas que “desaparecen” o se pierden también en conjunto, es decir, si al deteriorar un servicio determinado, se pierden al mismo tiempo otros servicios, como puede observarse, o al menos ser percibido socialmente, en la reducción del grupo de servicios de abastecimiento en el valle del Piedra[3] (Capítulo 5). En relación con esto, también sería interesante considerar en qué paisajes o bajo qué grado de intensificación del uso del suelo aparecen los compromisos o antagonismos entre servicios de los ecosistemas. Identificar estos umbrales es especialmente importante en la gestión de los agroecosistemas, para que puedan proveer el mayor número de servicios minimizando los antagonismos.

Comparar diferentes políticas de gestión implementadas para favorecer los servicios de los ecosistemas, y relacionar su funcionamiento con los factores socio-económicos del ámbito de aplicación para identificar en qué situaciones estas políticas podrían ser aplicadas a otros contextos. Para ello, habría que comparar en primer lugar diferentes políticas implementadas en un mismo tipo de paisaje y a continuación comparar entre diferentes paisajes o ecosistemas. Este trabajo requeriría de una extensa red de colaboración para recopilar datos de diferentes casos de estudio, o bien podría implementarse mediante modelos de simulación como los modelos basados en agentes (ABM, por sus siglas en inglés).

Comprobar que las directrices propuestas en esta tesis doctoral para la valoración social de los servicios de los ecosistemas son válidas para otros ecosistemas similares o no al de este trabajo. De esta manera podrían definirse de manera consistente los métodos más adecuados para la valoración social de los servicios de los ecosistemas. Esto reforzaría la utilidad del enfoque social frente al uso de métodos econométricos en la evaluación de los servicios de los ecosistemas por parte de la sociedad. Además, evaluar iterativamente los servicios de los ecosistemas por parte de la sociedad permitiría comparar si las preferencias sociales están relacionadas con el uso presente y pasado de los servicios, y por tanto cambian con el paso del tiempo (Oteros-Rozas et al. 2013). De esta manera se podría estimar la percepción social sobre el flujo temporal de servicios de los ecosistemas y comprender el cambio de las preferencias sociales a lo largo del tiempo.

Estimar los flujos de servicios de los ecosistemas a través del espacio y del tiempo, es decir evaluando los servicios de los ecosistemas a diferentes escalas espaciales y temporales. Considerar esta información permitiría estimar, por ejemplo, la distribución de los servicios de los ecosistemas en un espacio determinado a lo largo del tiempo, la cantidad de servicios disponibles en un lugar en un momento determinado, o las tendencias en la dirección de los flujos de servicios derivados del cambio del uso del suelo. El potencial de este campo es muy amplio, ya que se puede aplicar desde la escala local a la global. Por ejemplo, a escala local permitiría demostrar, como se ha comentado anteriormente, que los servicios de los ecosistemas, al igual que suelen aparecer agrupados (Raudsepp-Hearne *et al.* 2010), también desaparecen en grupos. A escala global otra posible aplicación sería, por ejemplo, trazar el flujo mundial de servicios de los ecosistemas y sus tendencias, revelando los principales productores y consumidores de cada servicio. Esta herramienta serviría para plantear políticas a nivel global que aseguren el acceso a los servicios de los ecosistemas de una manera más equitativa.

Profundizar en el efecto de las relaciones de poder en el flujo de servicios de los ecosistemas. Para ello, es necesario conocer el rol que desempeña cada agente social en el socio-ecosistema y su relación con el flujo de servicios de los ecosistemas. De esta manera, se podría estimar la pérdida o incremento de servicios de los ecosistemas a su paso por cada uno de los intermediarios en el flujo de servicios, y descubrir los factores clave que condicionan la distribución de servicios de los ecosistemas. Incorporando la identificación y análisis de las relaciones de poder al estudio de los flujos de servicios de los ecosistemas se lograría un conocimiento más profundo de los mecanismos que regulan el funcionamiento de los socio-ecosistemas. Esto permitiría descubrir, por ejemplo, los sumideros (tanto ecológicos como sociales) que modifican el flujo de servicios de los ecosistemas a la sociedad. Para desarrollar esta línea de investigación sería necesario incorporar al estudio de los servicios de los ecosistemas las perspectivas de la ecología política y de la antropología ecológica sobre la gestión y gobernanza de los recursos naturales.

Incorporar la conceptualización de los servicios de los ecosistemas en función de los derechos de propiedad y la importancia de la gestión de los bienes comunes (Lant et al. 2008). Para ello, habría que determinar el papel del uso del suelo y los derechos de propiedad en la posición que ocupa cada servicio de los ecosistemas dentro de un gradiente de exclusión y rivalidad. De esta manera se podría identificar el sistema de gobernanza de los socio-ecosistemas que permite un acceso más igualitario a los servicios de los ecosistemas y precisar en qué medida la gobernanza de los servicios de los ecosistemas determina su resiliencia. Es decir, comparar sistemas de gobernanza basados en un único o varios grupos de agentes de interés que usan y gestionan un servicio determinado y comprobar su contribución a potenciar el servicio o a agotarlo.

Reconocer el efecto positivo de ciertas acciones humanas en el mantenimiento del flujo de servicios de los ecosistemas para una mejor comprensión del funcionamiento de los socio-ecosistemas. Por ejemplo, la restauración de riberas favorece la recuperación de la estructura y funciones de los ecosistemas (Moreno-Mateos *et al.* 2012), la conservación de acequias mejora los servicios de soporte hídrico (Escalera-Reyes 2006), y el pastoreo contribuye a la diversidad de pastos y prevención de incendios (Ruiz-Mirazo 2012). Incorporar la labor positiva de la acción humana en la conservación de los ecosistemas y sus servicios podría motivar a la sociedad a colaborar en el cuidado de los ecosistemas, involucrándose en acciones que mejoren los servicios de los ecosistemas, como evitar la contaminación de las masas de agua, reducir la pérdida de suelo y el respeto a los seres vivos.

Finalmente, todavía queda mucho para lograr un *conocimiento holístico del funcionamiento de los ecosistemas* y de los factores que controlan cada uno de los flujos que relacionan a sus componentes. En este sentido, el marco de los servicios de los ecosistemas podría extenderse para considerar el efecto positivo y negativo que cada uno de los componentes de los ecosistemas ejercen sobre los demás, tanto a corto como a largo plazo. Numerosos autores argumentan que la degradación del medio ambiente puede derivarse del concepto de separación entre naturaleza y sociedad (Hansson and Wackernagel 1999; Barnaud and Antona 2014b; Escalera Reyes 2011), es decir, que el desacoplamiento de la sociedad respecto del ecosistema en el que vive ha hecho olvidar a las personas que su bienestar, e incluso su supervivencia, depende del funcionamiento de ecosistemas que tienen recursos limitados y, en muchos casos, insustituibles. Un conocimiento holístico del funcionamiento de los ecosistemas quizás contribuiría a evitar su degradación por desconocimiento de las interacciones que les afectan.

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[1] Funcionamiento de los ecosistemas que se percibe como negativo para el bienestar humano (Lyytimäki and Sipilä 2009).

[2] Modelo conceptual propuesto por (Haines-Young and Potschin 2010), donde se representan los servicios de los ecosistemas como un flujo desde el ecosistema al bienestar humano. Este modelo ha sido gradualmente modificado para incorporar nuevas aportaciones del campo de los servicios de los ecosistemas (de Groot *et al.* 2010; Kandziora *et al.* 2013; Martín-López *et al.* 2014), como la incorporación de los procesos sociales en el paso de "servicio" a "beneficio" (Spangenberg *et al.* 2014). Nuestra propuesta consiste en afinar este paso identificando las interacciones entre servicios de los ecosistemas y entre agentes sociales de interés que median y pueden impedir el acceso de las personas a los servicios de los ecosistemas.

[3] En esta zona, el abandono de la agricultura y la ganadería ha reducido no sólo la producción de alimentos sino también los recursos genéticos (variedades locales), el uso de plantas medicinales, la trasmisión de conocimiento local y las relaciones sociales (al no existir tantos bienes comunales que mantener en conjunto). Por otra parte, nuevos estilos de vida en la zona

han propiciado la aparición de un nuevo grupo de servicios culturales, como la educación ambiental y la investigación, la relajación y la recreación. Estos servicios fueron identificados tanto por los residentes permanentes como por los temporales (Felipe-Lucia 2012; ver Apéndice).

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CAPÍTULO 9. CONCLUSIONES

1. En la llanura de inundación del río Piedra, existe una alta relación entre una elevada provisión de servicios de los ecosistemas y mayor conservación de la biodiversidad a escala de paisaje, ya que cinco de los seis índices de diversidad estimados se correlacionaron positivamente con tres o más servicios de los ecosistemas. Sin embargo, a escala de parcela, la correlación entre servicios de los ecosistemas y biodiversidad depende del uso del suelo, siendo los hábitats naturales y semi-naturales los que proporcionaron más servicios y albergaron más diversidad que los usos de suelo cultivados.

2. La interacción entre los servicios de los ecosistemas y la diversidad vegetal depende del indicador utilizado para estimar la biodiversidad. La mayoría de los índices de diversidad vegetal se correlacionaron positivamente con los servicios de provisión de hábitat y educación ambiental y negativamente con la provisión de alimentos, pero estas interacciones también variaron en los diferentes usos del suelo.

3. La importancia relativa de cada uso del suelo en la provisión de servicios de los ecosistemas varía en función de la escala espacial a la que las medidas y los análisis se efectúan. Es decir, la cantidad de cada servicio de los ecosistemas depende de la provisión de servicio por unidad de superficie y de la superficie total que ocupa cada uso del suelo. En la llanura de inundación del río Piedra, los bosques de ribera proporcionan mayor cantidad de servicios de los ecosistemas a escala de parcela, mientras que a escala municipal y de paisaje los cultivos de cereal aportaron mayor cantidad de servicios debido a que ocupan la mayor extensión de terreno.

4. Conservar una pluralidad de usos del suelo es fundamental para proporcionar variedad de servicios de los ecosistemas; por ello, los bosques de ribera de las llanuras de inundación deben ser conservados y restaurados, ya que éstos son los ecosistemas que aporta la mayor biodiversidad a las llanuras de inundación y mejoran la provisión de los servicios de regulación, soporte y culturales.

5. La escala espacial de análisis condiciona el alcance de las interacciones (antagonismos o sinergias) entre servicios de los ecosistemas. De manera que no se puede concluir una única escala relevante para analizar las interacciones entre servicios de los ecosistemas. Por tanto, las políticas de gestión deben ser adaptativas, es decir, gestionando las llanuras de inundación a escala de paisaje, pero siendo capaces de adaptar medidas específicas para cada tipo de uso del suelo.

6. Identificar las causas de antagonismos o compromisos entre servicios de los ecosistemas es fundamental para aplicar medidas específicas que los reduzcan. En la llanura de inundación del río Piedra se han identificado antagonismos cuando los servicios de los ecosistemas eran mutuamente incompatibles en un uso de suelo determinado, mientras que en otros casos la provisión de servicios de los ecosistemas depende de decisiones de gestión sobre el uso del suelo.

7. Para llevar a cabo la valoración social de los servicios de los ecosistemas es necesario diferenciar: i) los servicios culturales del enfoque social, y ii) el enfoque social de la valoración económica basada en preferencias sociales. Además, se recomienda: 1) conocer los flujos de servicios comparando la percepción de servicios a través del espacio y el tiempo, 2) incluir agentes de interés de todos los rangos sociales y agruparlos según sus características sociales y su uso del ecosistema, 3) evaluar los servicios de los ecosistemas usando la doble vía de identificación y orden de preferencias, insistiendo en que los participantes identifiquen los servicios de los ecosistemas sin restringir las opciones a una lista dada y valorando cada servicio por cada uno de los beneficios que proporcione.

8. Los servicios de los ecosistemas no benefician por igual a la diversidad de usuarios potenciales debido a la existencia de relaciones de poder que influyen en las interacciones entre servicios de los ecosistemas y entre agentes sociales, modificando el flujo de servicios de los ecosistemas. La metodología empleada es útil para identificar servicios de los ecosistemas claves que determinan la provisión de otros servicios, servicios de los ecosistemas importantes para cada grupo de interés y servicios de los ecosistemas gestionados por un único grupo de interés, destacando el poder de este grupo para controlar el acceso y uso de los servicios de los ecosistemas.

9. En las relaciones de dependencia entre servicios de los ecosistemas de las llanuras de inundación del río Piedra destaca la importancia del uso y la gestión de los servicios de soporte y de regulación intermedios, pues son esenciales para la provisión de servicios fundamentales para el socio-ecosistema, como los de abastecimiento y los culturales.

10. La evaluación de los servicios de los ecosistemas es una herramienta útil para la gestión de los ecosistemas, especialmente, de las llanuras de inundación, porque permiten analizar el socio-ecosistema desde múltiples perspectivas en vez de desde un enfoque sectorial y detectar interacciones entre diferentes agentes sociales de interés.

CONCLUSIONS

1. *In the River Piedra floodplain, greater ecosystem services provision enhances biodiversity conservation at landscape scale. Our results showed that five diversity indexes were strongly correlated to three or more ecosystem services each one. However, at patch scale, correlations between ecosystem services and biodiversity changed according to the land use type. Natural or semi-natural habitats provided more services and hosted greater diversity than cultivated land use types.*

2. *Correlations between ecosystem services and plant diversity changed according to the indicator used to assess biodiversity. Most plant diversity indexes were positively correlated to habitat provision and environmental education, whereas food provision was negatively correlated to all diversity indexes. However, these interactions also changes across land use types.*

3. *The relative importance of each land use type in supplying ecosystem services changes depending on the spatial scale at which measurements and analyses are done. Thus, the amounts of each ecosystem service supplied depend on both the service supply per unit area and the total area occupied by each land use type. In the River Piedra floodplain, riparian forest supplied the most service of any land use type at the patch scale, but dry cereal croplands provided the most services across the municipality and landscape because of their large area.*

4. *Preserving a mixture of land use types is critical to providing a mixture of services. Thus, existing riparian forests should be preserved and restored across the floodplain as they are hotspots for floodplain biodiversity and enhance the supply of regulating, supporting, and cultural ecosystem services.*

5. *The spatial scale of the analyses conditions the interactions (synergies and trade-offs) among ecosystem services. Thus, there is no single relevant scale for analysing ecosystem services interactions. Therefore, policies might be adaptive, i.e. able to manage floodplains at the landscape scale while accommodating specific measures for each land use type.*

6. *Identifying drivers of ecosystem services trade-offs is key to applying specific measures to reduce them. In the River Piedra floodplain, we identified that some trade-offs were originated when services were mutually incompatible within a given land use, whereas the provision of others depended on land-management decisions within a land-use type.*

7. *For the social valuation of ecosystem services we need to distinguish: i) the cultural services from the social approach and ii) the social approach from the economic valuation based on social preferences. Additionally, we suggest: 1) understand ecosystem services flows by comparing ecosystem services preferences across time and space, 2) include a variety of stakeholders from all social ranges, grouping them according to their social characteristics and their use of the ecosystem, 3) evaluate ecosystem services via both identification and ranking, insisting that stakeholders nominate ecosystem services without listing constraints and separately evaluating the different benefits each ecosystem service supply.*

8. *Ecosystem services do not equally benefit the diversity of potential users due to power relationships mediating both ecosystem services and stakeholders interactions and modulating the flow of ecosystem services. Our analyses were useful to detect: i) keystone ecosystem services that determine the provision of other ecosystem services, ii) relevant services for each stakeholder group, iii) the ability of stakeholders for managing each service and their implications in other ecosystem services, and iv) power asymmetries between stakeholders derived from their capacity for managing ecosystem services*

9. *The dependency relationships between ecosystem services in the River Piedra floodplain highlighted the importance of the use and management of intermediate supporting and regulating services, as they are essential for the provision of final provisioning and cultural services for the socio-ecosystem.*

10. *Ecosystem services are a useful tool for environmental management, especially in agricultural floodplains, as they enable us to analyse the socio-ecosystem from multiple approaches rather than from a single point and identifying stakeholders interactions.*

APÉNDICE 1. FOTOGRAFÍAS

Créditos María R. Felipe Lucia salvo indicación expresa.

Recorrido fotográfico por la llanura de inundación del río Piedra

Vista aérea

Rueda de la Sierra. Uno de los tres "nacederos" del río Piedra.

Embid. Épocas secas (izquierda) y húmedas (derecha).


Torralba de los Frailes. Hoces del río Piedra a la altura del molino. Aspecto del cauce del río en épocas secas (izquierda) y húmedas (derecho).


Aldehuela de Liestos. Boca de las Hoces del Piedra.

Entre Aldehuela de Liestos y Cimballa.

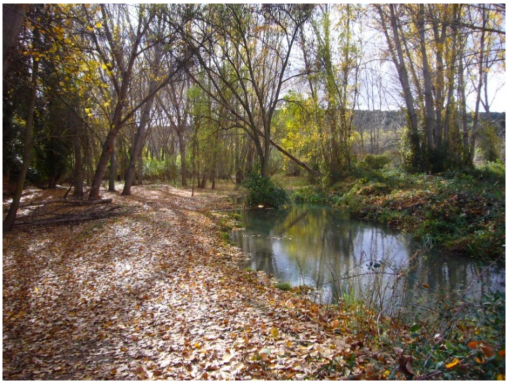
Cimballa. Ojos del río Piedra.

Cimballa. Vista de las riberas restauradas.

LLumes. Choperas y barbechos

LLumes. Cascada de la Calderera.

Nuévalos. Salto de la Requijada (*Gobierno de Aragón*)



Nuévalos. Paraje de Los Argadiles.

Nuévalos. Monasterio de Piedra (*FJ Ríos, Panoramio*)


Río Piedra a su paso por Nuévalos

Vista de Nuévalos


Embalse de la Tranquera en su capacidad máxima y mínima.


Carenas. Frutales y cultivos abandonados.


Castejón de las Armas. Casco urbano

Trabajo de campo




Transectos de vegetación en Embid y Nuévalos: Blanca, Pedro, María, Adrià.




Recogiendo muestras de suelos…y de la huerta: María, Carlos, Laura, Ricardo.

Colocando Ibuttons: Félix, Pedro.

Valorando los bosques de ribera: Paco.

Procesando muestras en el laboratorio: María.

Haciendo el 'juego de fichas' para la valoración social de los servicios de los ecosistemas.

Taller participativo para identificar servicios de los ecosistemas.

APÉNDICE 2. PRINCIPALES RESULTADOS Y BREVE DISCUSIÓN DE LA VALORACIÓN SOCIAL DE LOS SERVICIOS DE LOS ECOSISTEMAS DEL VALLE DEL RÍO PIEDRA

Introducción

Este apartado complementa los resultados derivados del caso de estudio referido en los capítulos 5 y 6, que puede encontrarse íntegramente en *Social dimension of Ecosystem Services: the case of river Piedra's valley* (Felipe-Lucia 2012). En primer lugar, se resumen las distintas opiniones de los agentes sociales de interés sobre el estado actual del río, sus causas y soluciones, subrayando los tópicos más recurrentes. Después se ofrece una síntesis de los comentarios sobre los servicios con los que el ser humano contribuye con el ecosistema. A continuación se resumen los principales servicios de los ecosistemas y del ser humano identificados en un taller en el que participaron dos representantes de cada grupo de interés. Finalmente, se discuten brevemente los resultados obtenidos.

Resultados

Estado actual del río, causas y soluciones

El 40.6% de los comentarios de los agentes sociales de interés entrevistados hicieron referencia al mal estado del río debido a la existencia de maleza en las orillas y ramas cruzadas en el cauce, que se considera "estar sucio", así como por las orillas derrumbadas en muchos puntos del río, que se entienden como un gran riesgo de inundación para los cultivos de cereal y las huertas. Las causas de este mal estado fueron relacionadas por los entrevistados con la falta de trabajos de mantenimiento del río (limpieza de maleza en las orillas y retirada de ramas cruzadas en el cauce), la incompetencia de las instituciones, las solicitudes de limpieza denegadas, el abandono agrícola y éxodo rural, las quejas y denuncias de los "ecologistas" y las agresivas limpiezas que se llevaban a cabo antiguamente. Las soluciones propuestas por ellos fueron: una mejora "ecológica" basada en la limpieza de la maleza (por tala o quema controlada) y/o el dragado del río y la recuperación de los cajeros de las orillas.

El segundo aspecto más destacado (12.5% de los comentarios) fue tanto la mala calidad de agua, como el buen estado del río. Por una parte, la mala calidad del agua se considera causada principalmente por la falta de una planta de tratamiento de aguas residuales en la piscifactoría de Cimballa y en los pueblos, que vierten directamente al río. Este hecho empeora durante los meses de verano, en los que la población se duplica y el flujo del agua disminuye, reflejándose en el mal color y olor de las aguas del río. La principal solución propuesta sería el tratamiento adecuado de las aguas residuales de la piscifactoría, así como una mejora en el sistema de captación de agua para, también con ello, contribuir a mejorar los caudales y la calidad del agua en el río.

Para algunos entrevistados este problema se relaciona más con el bajo caudal de agua, atribuido a la sobreexplotación de los pozos y a la extrema regulación del caudal en la presa después del período de regadío, que provoca cambios abruptos en el caudal del río, derivando en un mal estado ecológico del río. Para otros entrevistados, principalmente habitantes de aguas arriba de la presa, esto también se debe a la intensificación de la agricultura de regadío que conlleva el uso legal de productos químicos como fertilizantes y pesticidas. Sin embargo, los científicos y técnicos consideraron el estado ecológico del río Piedra intermedio, cuyas causas –a pesar de que el río Piedra recibe poca presión humana y la agricultura y ganadería es bastante extensiva– pueden asociarse con la falta de un corredor de vegetación en las orillas y/o de plantas de tratamiento de aguas residuales que reduzcan el efecto de los vertidos de los municipios, la piscifactoría, y los fertilizantes utilizados en la agricultura. La crisis económica es otro factor que retrasa la implementación de soluciones, como la restauración ecológica integral basada en un estudio exhaustivo y en consonancia con planes de desarrollo rural para esta zona.

Por otra parte, el buen estado del río se percibe únicamente por los agentes de interés externos y por aquellos que viven aguas debajo de la presa, y lo explican principalmente por el abandono rural, la agricultura extensiva, el éxodo rural y la pérdida de la ganadería y por la clarificación del agua tras su paso por el embalse. Esta opinión también fue apoyada por los entrevistados que consideran que la principal función del río es transportar agua para los regadíos. Sin embargo, la belleza estética se atribuye principalmente a las cascadas, al sonido del agua y al paisaje en general, por lo que alguno de los agentes de interés propone la recuperación de la maximización estética del paisaje como objetivo.

Finalmente, la desaparición de la pesca se relaciona principalmente con el dragado acometido hace unos 20 años que cambió la composición del lecho de río de gravas a lodos negros. Las soluciones propuestas para paliar este problema fueron la reducción del uso de productos químicos en la agricultura y la reintroducción de la trucha común. Además, por un parte se percibe que los recursos están desprotegidos, por el riesgo que supone el agotamiento del acuífero del Piedra, y por otra parte, que los recursos están infrautilizados, en referencia al mal estado del patrimonio tradicional relacionado con el uso del agua, como las norias. Ambos problemas fueron señalados por habitantes temporales de la parte media del valle. Las soluciones propuestas fueron la protección del acuífero e inversión en el entorno, y la promoción del ecoturismo, respectivamente.

Percepción de cambio en los servicios del ser humano

Catorce entrevistados aportaron información del cambio producido en los últimos 50 años en los servicios con los que el ser humano contribuye con el ecosistema. La mayor parte de los comentarios hicieron referencia a una pérdida de los servicios que aporta el ser humano. Por ejemplo, los habitantes aguas arriba de la presa remarcaron que el servicio que más había disminuido era la *limpieza de orillas y del lecho del río*. Otros percibieron una reducción en *reponer las orillas* y en *cuidar las especies del río*. Los servicios de *cuidar de las avenidas* y *restaurar la arquitectura* se perciben sin cambios y la *limpieza de basuras* se ha incrementado.

Comparación de servicios de los ecosistemas y servicios del ser humano en un taller participativo

En este taller, los participantes –dos representantes de cada grupo de interés– propusieron y valoraron los servicios de los ecosistemas y los services del ser humano más importantes tanto en la actualidad como hace 50 años. El servicio más importante proporcionado por los ecosistemas para los participantes fue el *desarrollo económico*, tanto hace 50 años como en la actualidad. Otros servicios de los ecosistemas importantes hace 50 años fueron el actuar como *motor social*, y en menor medida, la *calidad del medio ambiente* y el *conocimiento del medio*. En la actualidad, otros servicios que proporciona el ecosistema son *calidad de vida*, *consumo de agua* y *conocimiento sobre el agua*. Los principales servicios con los que el ser humano contribuía hace 50 años eran *cuidar del estado del valle en general* y *relaciones sociales y de convivencia*, y en segundo lugar, *difusión y prestigio de los productos del valle*. En la actualidad el ser humano contribuye principalmente con un dis-servicio, la *contaminación*, pero también con *conocimiento medioambiental*, *nuevos usos para el desarrollo económico y social*, *actuaciones de mejora*, e indirectamente con una *inversión económica forzada* a través de impuestos. Las causas de estos cambios fueron relacionadas con los cambios en la mentalidad de la sociedad en general así como en la población de estudio.

Discusión

Las causas de la percepción de cambio en la provisión de servicios de los ecosistemas en los últimos 50 años se relacionan con el cambio general de la sociedad española durante este periodo. De un valle basado y dependiente de la provisión de servicios de abastecimiento, se ha pasado a un valle basado, pero también dependiente, de la existencia de servicios culturales, fundamentalmente relacionados con las actividades recreativas. Este caso de estudio es un reflejo de cómo nuestra sociedad ha reemplazado su propia seguridad alimentaria por una dependencia total del turismo y de las subvenciones externas (p.ej. las ayudas de la Política Agraria Común europea). Del mismo modo, la antigua provisión de servicios culturales (basada en las relaciones sociales alrededor de la vida en el río, como lavar, fregar, bañarse, y reunirse) ha cambiado hacia un turismo de naturaleza organizado (rutas senderistas señalizadas, guías de naturaleza, visitas guiadas).

En general, se observa que los servicios de los ecosistemas utilizados directamente en la actualidad (es decir, los servicios culturales y de abastecimiento) se perciben como más importantes que los servicios que no se utilizan directamente o que son más difíciles de percibir (como los de soporte y regulación). Además, se observan ciertas diferencias en la percepción del estado del río y los servicios de los ecosistemas según la localización de los entrevistados con respecto al embalse de la Tranquera. La población aguas abajo de la presa tiene una visión más positiva del río que los habitantes aguas arriba. Éstos están más aislados de los municipios principales y su economía está basada principalmente en la productividad de los cultivos, mientras que los habitantes de los municipios próximos al cauce o de aguas abajo de la presa están más próximos a otros municipios principales y trabajan principalmente en el sector turístico. Esto indica que los habitantes aguas arriba de la presa tienen menores conexiones con las redes sociales, resultando en una sociedad más reducida que los municipios mejor conectados (es decir, los de aguas abajo). Esta diferencia también se observa en el conocimiento de la zona. Por ejemplo, los científicos, técnicos y trabajadores del sector turístico cuentan con una visión general de la cuenca, mientras que los que se dedican al sector primario suelen referirse siempre al tramo del río más próximo.

El concepto de servicios con los que el ser humano contribuye a los ecosistemas está comenzando a recibir más atención desde la academia (Huntsinger and Oviedo 2014). Sin embargo, todavía es necesario mucho debate para establecer una definición y tipología. Por ejemplo, en nuestro caso de estudio, las orillas canalizadas mediante bloques de piedra fueron consideradas para algunos agentes de interés como un elemento de la arquitectura tradicional de importancia cultural, mientras que para otros constituye un obstáculo para el buen funcionamiento ecológico del río. En cualquier caso, la valoración de los efectos tanto positivos como negativos del ser humano en el ecosistema deberían ser considerados para una mejor comprensión de las dinámicas de los ecosistemas.

Referencia

Felipe-Lucia 2012. *Social dimension of Ecosystem Services: the case of river Piedra's valley*. Universidad Pablo de Olavide.

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“Cuenta una leyenda que todo lo que cae en las aguas de este río —las hojas, los insectos, las plumas de las aves— se transforma en las piedras de su lecho”

A orillas del río Piedra me senté y lloré

Paulo Coehlo

“Cuenta otra leyenda que en un día no muy lejano, las piedras y los limos de este río revivirán y se distribuirán proporcionando alegría y bienestar. Pero habrá que estar atentos, porque el cauce de este río también se puede secar para siempre si no se cuida con amor”

Anónimo

“Si queremos seguir disfrutando de nuestros ríos, bañándonos en ellos, paseando junto a ellos, o incluso bebiendo de sus aguas, debemos adoptar la perspectiva no dual. Debemos meditar en ser el río, para que podamos experimentar en nosotros los miedos y esperanzas de los ríos. Si no podemos sentir los ríos, las montañas, el aire, los animales, o al prójimo desde su propia perspectiva, los ríos morirán y perderemos nuestra oportunidad para alcanzar la paz”

Hacia la paz interior

Thich Nhat Hanh

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